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Addressing the land loss-fish production paradox

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ADDRESSING THE LAND LOSS-FISH PRODUCTION PARADOX

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To my late father, Gary Lewis, to my late uncle, David Lewis, and to my late grandmother, Shirley Lampe: for always encouraging me to wander and learn, my journey is only beginning.

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ABSTRACT

Saltmarsh loss is occurring at high rates in Louisiana (LA), but understanding the impacts that marsh degradation has had on historical abundance of estuarine nekton in Barataria Bay, LA is lacking. I first examined the differences between fishery independent and fishery dependent data as indices of relative abundance. Previous studies used landings data to evaluate the importance of marsh habitat (e.g. distance of marsh edge and area of intertidal marsh) to fisheries production, but for most species, landings and survey data showed differing patterns of abundance through time. These findings emphasize the importance of using survey data (not landings data) to conduct habitat-related analyses in Louisiana and elsewhere. Next, I investigated the influence of a suite of environmental and fishery related predictors on fishery independent catch-per-unit-effort (CPUE) and developed descriptive models for these relationships. The descriptive models show that abundance of estuarine nekton varied only marginally with marsh habitat related predictors. Using both parametric and non-parametric statistical analyses, I then tested previous hypotheses that explained the relationship between marsh habitat and fish abundance, in addition to exploring community level effects. Results indicated that marsh and marsh edge distance do not appear to be driving nekton abundance. However, differences were found when comparing the community structure of Barataria Bay from different time periods over the last 44 years. Finally, I developed an ecosystem model to test the influence of marsh loss on nekton abundance, while accounting for changes in salinity and trophic interactions. Results indicate that marsh edge accounted for only a small portion of historical variation in nekton abundance. While this study suggests the influence of marsh loss on fisheries may be less significant than once thought, the importance of protecting coastal saltmarshes remains vital to the health and prosperity of Louisiana's deltaic ecosystem.

CHAPTER 1.
GENERAL INTRODUCTION

1.1. Rationale

Over the past 200 years, exploitation of our ocean resources has risen significantly due to the advent of better technology (Myers and Worm 2003). The ability to fish longer and fish in areas historically difficult to access is compounded when one considers other factors that can affect the abundance and distribution of marine species. Climate change, differences in scientific analysis, and even mistakes in management can contribute to declines in fisheries (Beamish et al. 1999, Healey and Hennessey 2000, De Mutsert et al. 2008). In addition, marsh habitat loss and alterations are affecting fisheries in many regions throughout the world, and this issue has long been a topic of concern for scientists who focus their work in the northern Gulf of Mexico (Thomas 1998). Coastal Louisiana (LA) accounts for 60-80% of the nation's annual wetland loss (Boesch 1994). Fishes that inhabit these estuaries are not only important to recreational and commercial anglers, but are important as indicators of estuarine health (Whitfield and Elliott 2002). Researchers have long recognized the importance of estuaries as nurseries for species that live in estuary and continental shelf ecosystems (Gunter 1967, Nixon 1980, Boesch and Turner 1984, Baltz et al. 1993). In fact, one highly cited study directly related the area of intertidal marsh to the amount of penaeid shrimp yield (Turner 1977). But as fisheries landings in LA remained stable or increased in the face of marsh loss over the last 50-70 years, studies emerged to find mechanisms for this resiliency (Browder et al. 1985, Browder et al. 1989, Rozas and Reed 1993, Zimmerman et al. 2002). These studies, and many others, infer that the marsh edge effect, which changes as degradation continues through time, will lead to an optimum amount of edge interface thereby increasing the availability of marsh use by estuarine nekton.

Increased edge, is believed to mitigate the consequences of overall loss of marsh habitat. But as marsh degradation continues, the fragmented marsh begins to wash away, along with marsh edge. So over the short term, it is thought that species that utilize LA estuaries would most likely benefit from increased edge, but at the point where marsh becomes more than 50% open water, fish production (and therefore yields) would decrease. The question then becomes, where are we on the marsh degradation continuum? If this conceptual model holds true for LA marshes, then we must still be experiencing the benefits of increasing marsh edge interface since landings remain stable or are increasing. However, I found that the maximum linear distance of the marsh edge (hereafter edge) in Barataria Bay occurred in 1985. So it has been over 25 years since edge achieved its maximum, and since that time we have seen a steady decrease in both area of marsh and edge, concurrent with stable and increasing yields of important commercial and recreational fisheries. In the light of these findings, there is a need to determine whether or not edge and marsh area are important drivers of nekton abundance, and if not, seek to identify variables that explain variation in the long term abundance of the nekton community.

1.2. Ecosystem Description

The Mississippi River (MR) drains over 40% of the continental United States and is the 8th largest river system in the world (Turner and Rabalais 1991). The river carries sediments and nutrients that have helped create a productive deltaic ecosystem in Louisiana (Coleman et al. 1998). Many species of fish, invertebrates, birds, and marine mammals utilize the estuarine waters for at least some portion of their life history. While productive, Louisiana's coastal marshes are particularly fragile consisting of hundreds of miles of Holocene deltas built by sediment deposition from the MR and its distributaries. Human-induced changes within Gulf of Mexico estuaries include leveeing, channelization, impoundments, and diversions. Changes

were made within the estuaries to reduce the periodicity of natural flooding, to allow large ships to navigate inland, and to develop infrastructure for the oil and gas industry. The MR has been completely leveed since 1942, effectively stopping the natural cycle of delta lobe abandonment and creation. This, in part, has resulted in Louisiana accounting for most of the land loss in the lower 48 states (Barras et al. 2003). Specifically, Louisiana loses approximately 65 km^2 of wetland ecosystems per year. Marsh loss has been the most profound and unforeseen consequence since the dissociation of the river from its adjacent land. Current rates of geological subsidence show no sign of decreasing in southern Louisiana (Rozas and Reed 1993, Lane et al. 2007). Subsidence in turn, increases the rate of local sea level rise (SLR) when compared with eustatic SLR. The loss of marsh allows salt water to intrude higher and higher into the basin consequently affecting the distribution and abundance of important commercial and recreational species such as blue crab (*Callinectes sapidus*), penaeid shrimp and spotted seatrout (*Cynoscion nebulosus*).

1.3 Coastal Restoration

Saltmarshes serve many important functions such as providing protection from storms and improving water quality. Attempts to reintroduce water and sediment to coastal marshes, similar to that of a natural delta ecosystem, have brought about a number of restoration approaches. River diversions have been built in LA to help mitigate flooding and have been used to address salt water intrusion or when an oil spill occurs, the diversion can be opened to flush oil from the estuaries. The timed release of water from the primary river can both reduce potential flooding in now inhabited flood plains and allow for the upper reaches of the estuary to return to the more oligohaline and mesohaline conditions historically observed. In Barataria Bay, the Davis Pond Freshwater Diversion provides a controlled flow (~28 m^3/s) of freshwater

into the upper reaches of the estuary. When this water control structure was completed in 2002, the primary goal was to maintain a salinity of 5 PPT within the estuary, but recently plans have been made to adjust the flow to better mimic historic natural flooding that would have occurred if the MR had not been completely leveed (Wold 2013). It has been discussed, however, that using diversions as a way to reintroduce sediment and nutrients back into LA estuaries could be a way to help mitigate marsh loss in coastal LA (DeLaune et al. 2003, Day et al. 2007).

1.3. Dissertation Overview

In Chapter 2, I investigate the influence of a suite of environmental and fishery related explanatory variables on the relative abundance of six important nekton species in Barataria Bay, LA. I first investigate the null hypothesis that survey, landings, landings catch-per-unit-effort data (CPUE) can similarly depict the relative abundance for these species. I then fit a series of multiple linear regression models to test the null hypothesis that species abundance is not influenced by environmental or fishery related predictors. These analyses were used to inform and corroborate the analyses conducted in the next two chapters.

In Chapter 3, I test the null hypothesis that continued marsh loss in coastal LA will have no detrimental effects on fish abundance and community structure. Using the same species of interest from Chapter 2, I first regress survey, landings and landings CPUE data on area of marsh habitat and distance of marsh interface to compare patterns of abundance. I also use a non-parametric approach (tree regression) to determine if the decrease in marsh interface since the maximum value observed in 1985 had a noticeable impact on the relative abundance of the species of interest. Finally, I compare species biomass distributions over 44 years to determine if marsh edge loss has effected community structure within Barataria Bay.

In Chapter 4, I create a mass-balanced and time-dynamic ecosystem model using the Ecopath with Ecosim (EwE) software package (Christensen et al. 2008). This model tests the null hypothesis that marsh edge drives variation of nekton abundance over time, while concurrently accounting for other environmental factors and trophic interactions. The model is fit to historical survey data for commercially and ecologically important nekton species and the model is driven by salinity and marsh edge forcing data. Biomass model predictions are compared to observed biomass values in the model output.

I conclude in Chapter 5 with a summary of the major empirical findings from the statistical analyses and EwE model outcomes. I discuss both the theoretical and policy implications of this study. I then address the limitations and how these limitations can be improved upon in future research. Finally, I discuss improvements in the EwE modeling software and how these improvements could help to explain the paradox of land loss and fish production in coastal Louisiana.

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CHAPTER 2.
HISTORICAL ANALYSIS OF FISHERIES AND ENVIRONMENTAL DATA IN BARATARIA BAY, LOUISIANA, USA

2.1. Introduction

Fisheries landings in Louisiana (LA) have proven robust considering the many environmental and anthropogenic perturbations the coastal ecosystem has experienced (Chesney et al. 2000, Zimmerman et al. 2002, Cowan et al. 2008). While many commercially important species in LA can be considered a "yearly crop" because of their short life history, using landings data alone to determine the status of fisheries, and thus the health of the estuarine ecosystem can be misleading (De Mutsert et al. 2008). Recreational catch, on the other hand, has largely been ignored in the evaluation of the resiliency fisheries landings in the face multiple perturbations. In addition, LAs coastal ecosystems vary greatly from east to west in terms of land loss rates, salinity, dissolved oxygen, and freshwater input (among other factors). Many studies have aggregated data over all basins, which could potentially confound true patterns of abundance within this very dynamic ecosystem (Turner 1977, Browder et al. 1989).

Chesney et al. (2000) spoke specifically about data biases, discussing the resiliency in LA landings and then analyzing 21-years of fishery independent data to corroborate the findings in the fishery dependent data. This type of comparative analysis was made possible because the LA Department of Wildlife and Fisheries (LDWF) has amassed a large fishery independent database tracking the historical trends of estuarine nekton for over 40 years. Because of the cost prohibitive nature of fisheries independent surveys, landings data are often the only available measure of fish abundance. Fortunately, this situation is not the case in LA, where there is a long time series of data collected in each of LA's seven estuarine basis, using a variety of

sampling gears. I used these survey data, along with landings and landings catch-per-unit-effort (CPUE) data to test the null hypothesis that all three types of data cannot be used as indexes of abundance. Then, to test the null hypothesis that species abundance varies randomly through time, I developed descriptive models using environmental and fishery related predictors for six species of interest.

2.2. Methods

2.2.1. Study Area and Species Descriptions

The LA coast is separated into seven Coastal Study Areas (CSAs, Figure 2.1) to delineate the natural geographic changes that occur from east to west along the coast. This study focuses on CSA 3, which encompasses Barataria Bay, a 6280 km^2 sub-region of the Barataria-Terrebonne estuary system and member of the National Estuary Program (Nelson et al. 2002). Barataria Bay (Figure 2.2), isolated from the Mississippi River (MR; which is the easternmost boarder of the basin) since the 1940's, gets a majority of its freshwater input from rainfall and includes freshwater, brackish, and marine coastal marshes. The basin is bounded on the west by Bayou Lafourche, an abandoned main stem of the MR (Conner and Day 1987). In addition to rainfall, the Davis Pond Freshwater Diversion provides a controlled flow (~28 m^3/s) of freshwater into the upper reaches of the estuary.

The species chosen for this analysis were selected because of their importance as commercial and recreational fisheries and their putative dependence upon estuarine nursery areas (Zimmerman et al. 2002). The species are spotted seatrout (*Cynoscion nebulosus),* red drum (*Sciaenops ocellatus),* brown shrimp (*Farfantepenaeus aztecus),* white shrimp *(Litopenaeus setiferus),* blue crab *(Callinectes sapidus),* and Gulf menhaden (*Brevoortia patronus).*

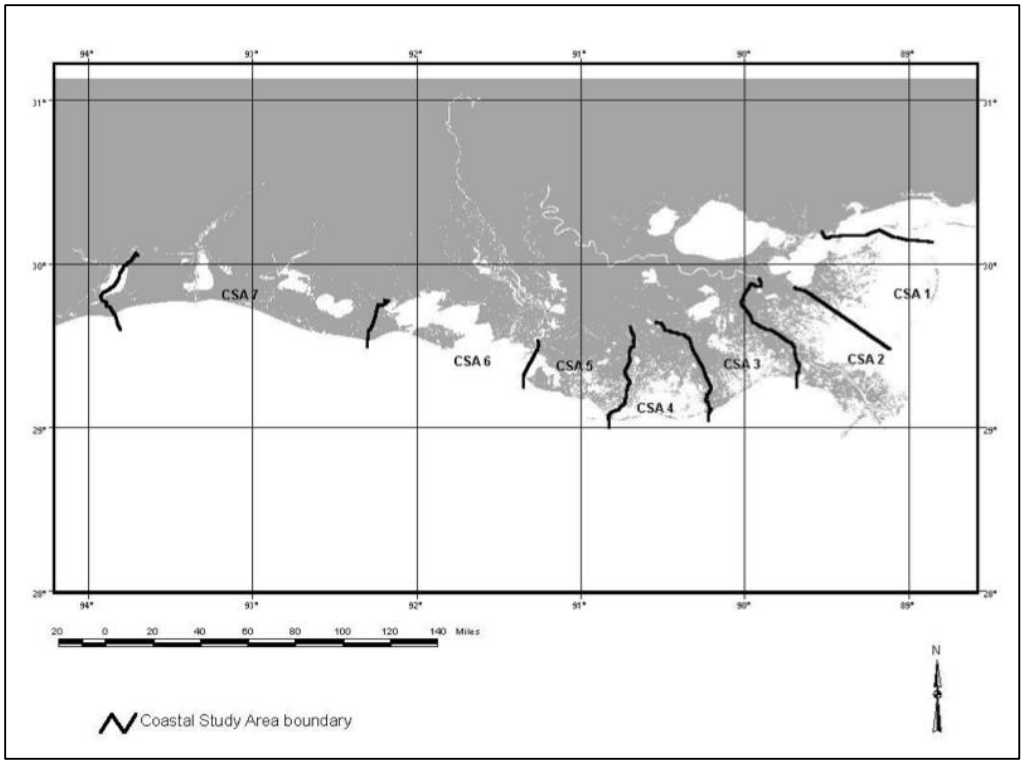

Figure 2.1. Louisiana's seven Coastal Study Areas. Barataria Bay is located solely within CSA 3.

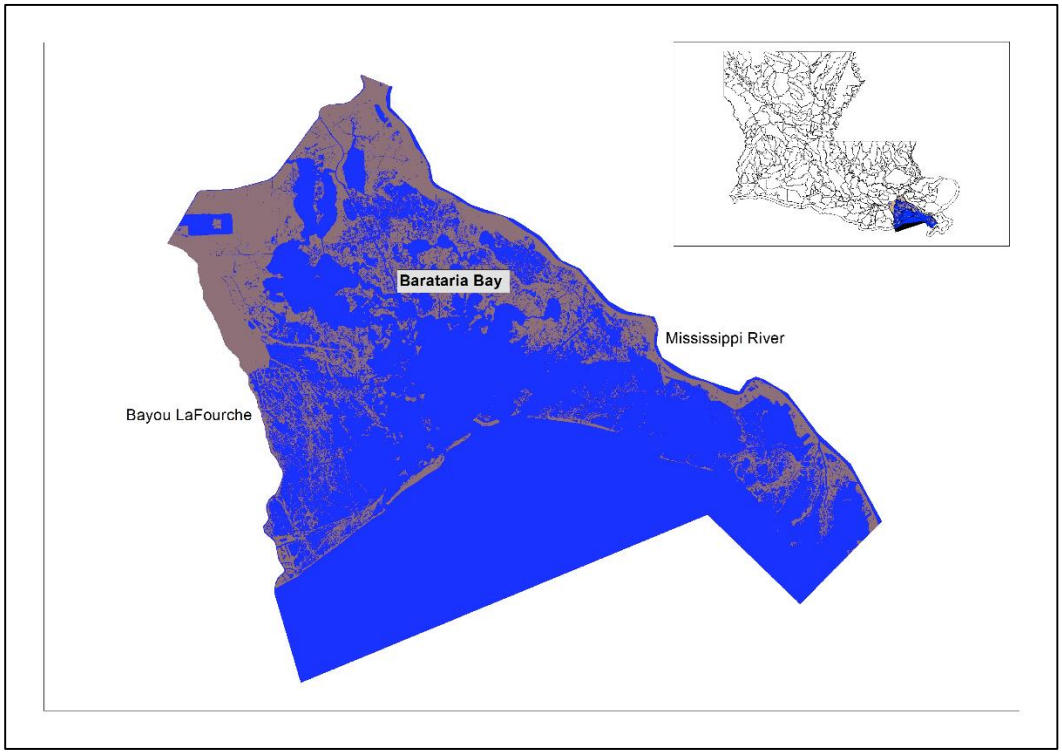

Figure 2.2. Barataria Bay, LA, USA, bordered on the east by the Mississippi River and on the west by Bayou LaFourche (Couvillion et al. 2011).

Spotted seatrout (SStrout) are found in coastal estuaries from the northern Atlantic coast (Cape Cod, MA) to the Gulf coasts of Mexico (Bay of Campeche) (McEachran and Fechhelm 2006). Spotted seatrout show a high degree of estuarine fidelity, and recent studies have corroborated earlier findings that most SStrout stay very the near estuaries that were their nursery grounds (Callihan et al. 2012).

Louisiana has the largest recreational harvest of SStrout in the U.S.; landings have been steadily increasing since the 1980s (Figure 2.3). Spotted seatrout have been fished commercially since the late 1800s, primarily as by-catch in the seine fishery and were later targeted with trammel nets and gillnets (Bowman et al. 1977). However in 1997, SStrout were designated as a game fish thus ending catches with commercial gears of any type.

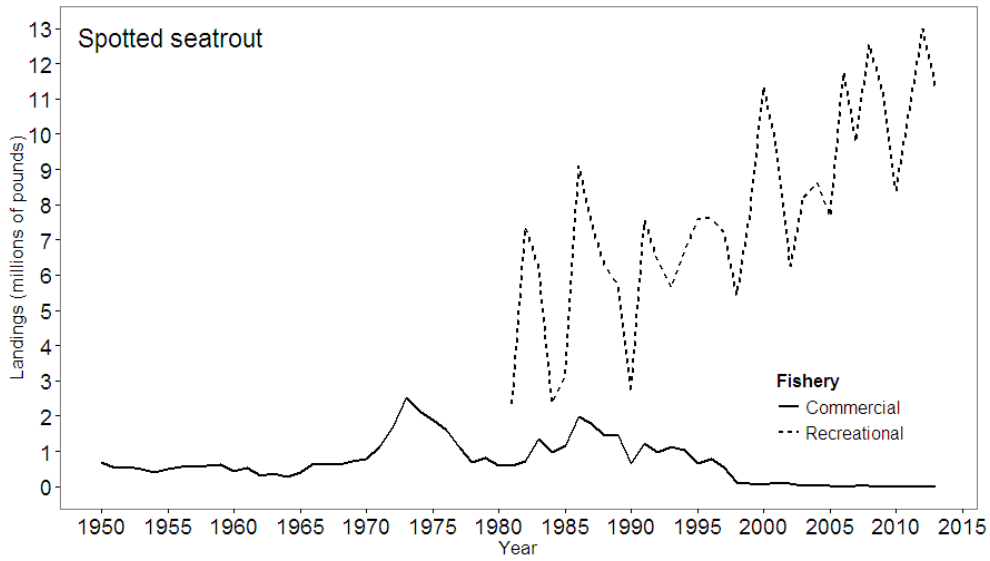

Figure 2.3. Louisiana commercial and recreational landings for spotted seatrout (*Cynoscion nebulosus*) from 1950 to 2013.

Red drum is an estuarine dependent species of sciaenid that ranges from Massachusetts to the northern Mexico (Murphy and Taylor 1990). Red drum enter estuaries as post-larvae or small

juveniles, moving offshore as adults between the ages of 3 and 6 (Murphy and Taylor 1990, Powers 2012). Adult red drum (RD) spawn near river inlets in the fall (Overstreet 1983), along the continental shelf, and in estuaries, exhibiting variation in spawning patterns (Holt et al. 1985).

Red drum commercial and recreational harvest changed significantly in the past 50-60 years (Figure 2.4). A majority of the harvest prior to the 1980's was commercial, with a dramatic increase in catch concurrent with the rise in popularity of "blackened redfish" mid-decade. The increased effort led to a substantial decrease in spawning stock biomass, which led to development of a Fishery Management Plan (FMP) that placed a moratorium on commercial catch in federal waters in 1987 (Powers 2012). Recreational catch of red drum also experienced a spike in landings in the mid-1980s and has steadily increased while commercial landings remain low (Figure 2.4).

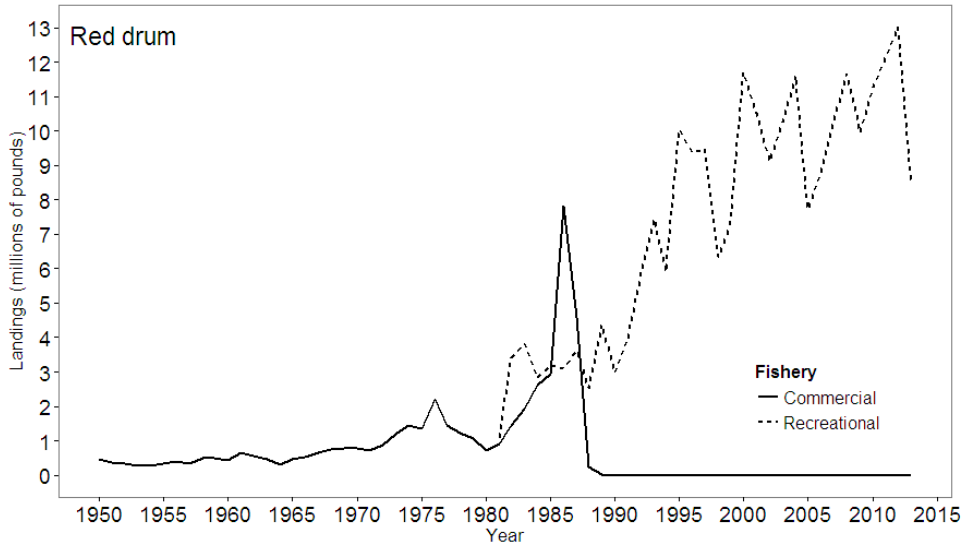

Figure 2.4. Louisiana red drum (*Sciaenops ocellatus*) commercial and recreational landings from 1950 to 2011.

Brown (BS) and white shrimp (WS) live no longer than a year and are managed accordingly in LA (Haas et al. 2001, Cowan et al. 2008). Adult shrimp of both species spawn offshore; brown shrimp larvae, as with other penaeids, pass through 11 life history stages over a 10- to 25-day period before transforming into post larvae that ultimately must be transported into an estuary by currents and tides, where the post-larvae settle and undergo rapid growth before again migrating offshore to spawn.

Louisiana's shrimp fishery lands the most shrimp by weight compared to anywhere else in the US and is second only to Texas for the most valuable shrimp fishery. Shrimp landings have experienced a discernible increase (WS) or have remained relatively stable (BS) since the 1970s (Figure 2.5). The stable or increasing landings are remarkable considering the reduction of fishing effort, a result of outside market pressure from the Asian shrimp industry, a rise in fuel prices, and a reduction in infrastructure after Hurricanes Katrina and Rita.

Blue crab (BC) range from the maritime provinces of Canada south to the Gulf of Mexico and the Caribbean Sea and likely inhabit a single estuary for their entire life history. To spawn, females seek out higher salinity waters, where recruitment and settlement of the larvae are dictated by tides and currents (Guillory et al. 2001). Commercial landings of BC have been recorded as far back as 1880; however little information regarding historical recreational landings has been found. There was an increase in commercial effort between the 1980s and 1990s resulting in an increase in landings (Figure 2.6).

Gulf menhaden (GuM) is an estuarine dependent species that is harvested as an annual crop in Louisiana. Gulf menhaden range from Cape Sable, FL to Veracruz, Mexico (Reintjes 1969) and can be found in inshore waters during the spring months, but in late fall, they can be

found far offshore. This euryhaline species spawns October through March, peaking sometime either in December or January (Vaughan et al. 2007). After hatching at sea, the larvae move into the estuaries and remain there until late fall when they migrate back offshore (Shaw et al. 1985).

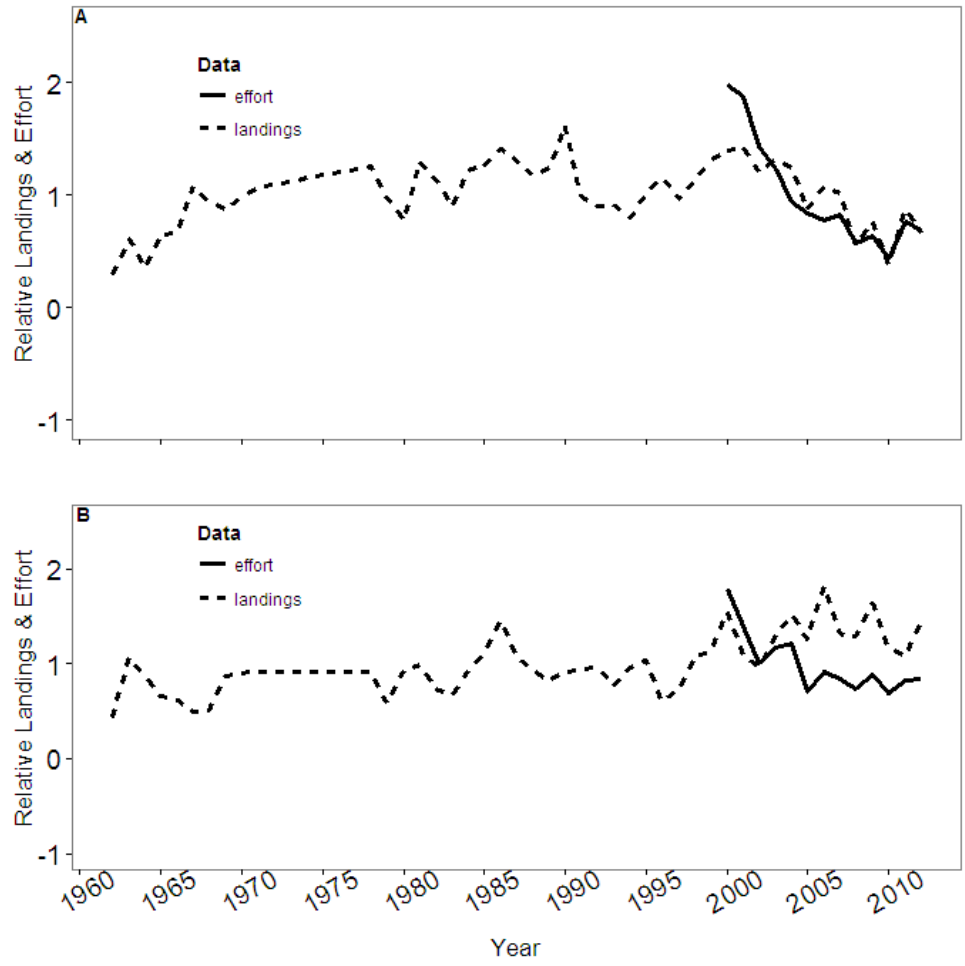

Figure 2.5. Panel A: Louisiana relative commercial landings from 1962-2012 for brown shrimp (*Farfantepenaeus aztecus*) (dashed line) and brown shrimp commercial effort from 2000-2012 (solid line); Panel B: Louisiana relative commercial landings for white shrimp (*Litopenaeus setiferus*) from 1962 to 2012 and white shrimp commercial effort from 2000 to 2012 (solid line).

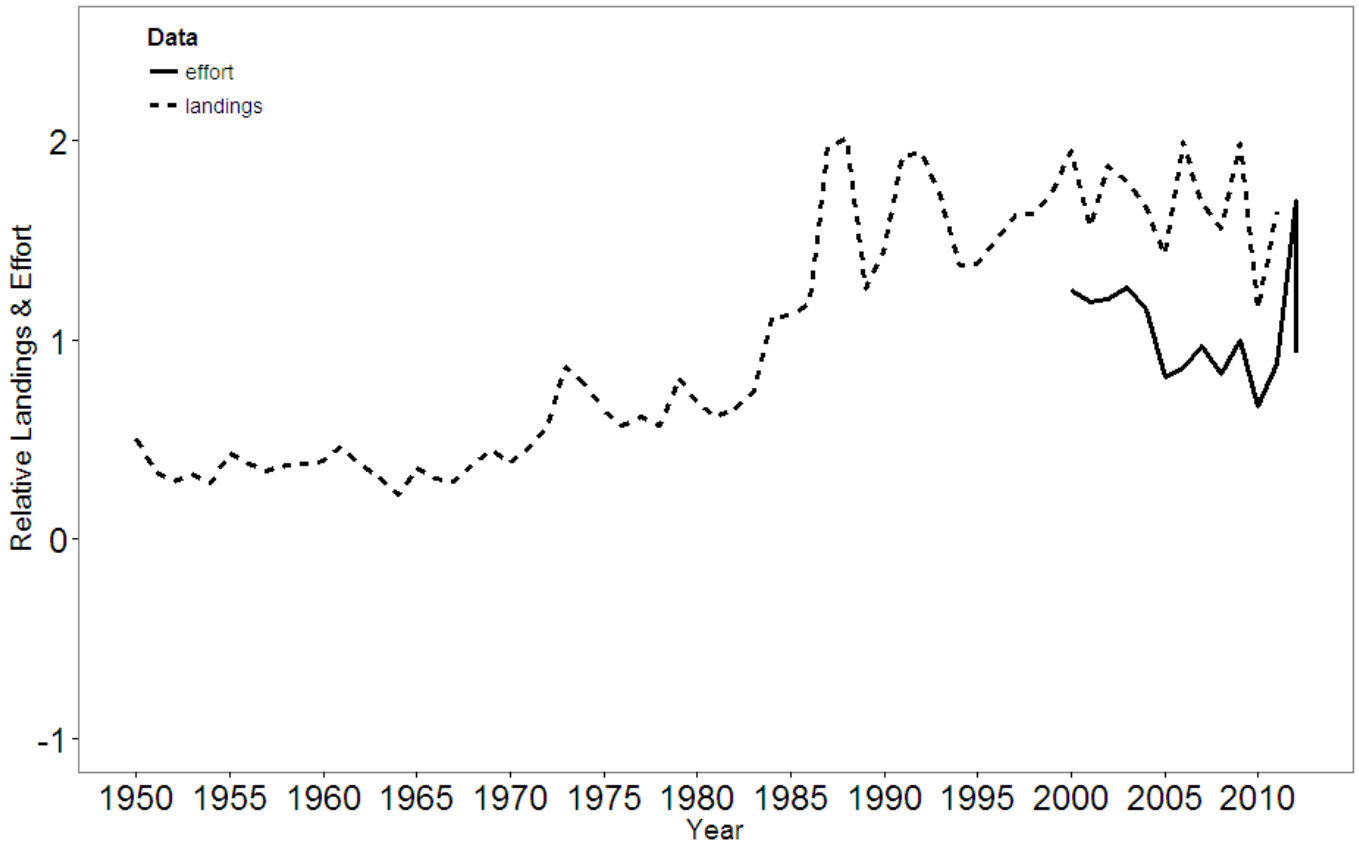

Figure 2.6. Louisiana commercial blue crab (*Callinectes sapidus*) landings from 1950 to 2012 and commercial effort from 2000 to 2012.

There are records documenting a GuM fishery in the late 19th century, though records are patchy at best (Nicholson and Schaaf 1978). During the 20th century GuM were subject to an extensive purse seine fishery, with landings steadily increasing through the mid-1980s, followed by a decline. In the last 5-10 years, landings have shown a steady increase (Figure 2.7). Louisiana alone harvests 92% of the annual GuM catch for the entire Gulf of Mexico fishery, emphasizing the economic importance in the harvest of this species (Vaughan et al. 2007).

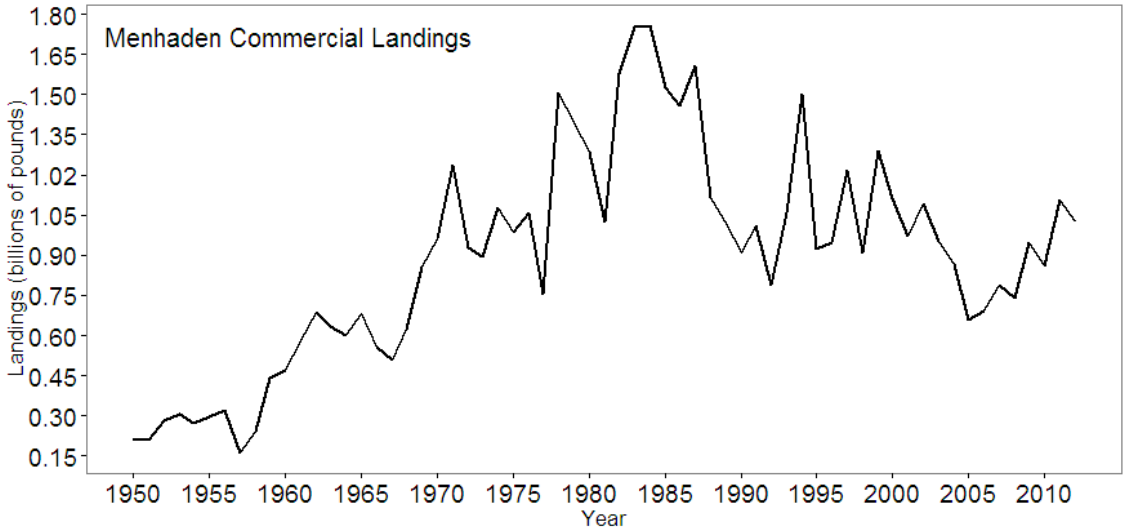

Figure 2.7. Louisiana Gulf menhaden (Brevoortia patronus) commercial landings from 1950 to 2012.

2.2.2 Sample Design and Data Collection

Louisiana continues a fishery independent monitoring program that began in 1966. The Louisiana Department of Wildlife and Fisheries utilizes different gear types to target certain species. With this a priori information I calculated the overall catch (combined over years) by gear type to determine which gear provided the highest sample numbers per species (Table 2.1). Although abundance collected may be higher with other gear types for some species such as RD and GuM, gillnets were chosen for RD so adult fish could be considered in the analysis and trawl gear was chosen for GuM because of outlier data found in the gillnet data for that species. To

that end, I utilized data collected by LDWF with either a 4.8 m. flat otter trawl or a 229 m. gillnet for this study, because in general, the highest number of species were caught with these gear types. Trawl sampling began in 1966 (data are sporadic during the first year of sampling, so this study used data beginning in 1967), while gillnet sampling began in 1986. Though abundance data were available until 2013, data were only analyzed until 2010 due to data limitations from other environmental variables (marsh edge and area of marsh). The total number of samples collected over all years for trawl gear and gillnet gear was 9044 and 4630, respectively.

Table 2.1. Overall catch (abundance) of all target species for all gear types during the period of record (POR) in Barataria Bay. Species of interest data for these analyses were obtained strictly from two gear types: trawl and gillnet. The catch numbers indicated with an (*) represent the gear by which each species' data were analyzed.

	Trawl	**Gillnet**	**Seine**	**Trammel**
Species\POR	**1967-2010**	**1986-2010**	**1986-2010**	**1986-2010**
Spotted seatrout	516	18169*	788	252
Red drum	240	1181*	1692	674
Brown shrimp	228779*	24	13731	3
White shrimp	75951*	3681	10904	6
Blue crab	38852*	3189	5001	233
Gulf menhaden	30179*	48489	71125	288

Trawl gear, (primarily used to obtain estimates of penaeid shrimp abundance and to predict the opening of shrimp season), is attached to a 1.27 cm. diameter nylon rope or stainless steel line and bridle that is approximately 2 times the length of the trawl itself. The towline is typically 4 times the maximum water depth. Trawl time is ten minutes at a constant speed (~3.0 knots) in a weaving or circular pattern to allow for the prop wash to pass on the sides of the trawl. The body of the trawl is made of 1.9 cm. bar mesh No. 9 nylon mesh and the codend is

constructed of 1.9 cm. bar mesh knotted 0.45 kg tensile strength nylon and is about 1.4 m. long. The organisms collected in the trawl are identified to species, counted, and a maximum of 50 of each taxa are measured in 5 mm intervals. LDWF collects trawl samples every other week from November to February and weekly from March to October. For the purposes of this study, one ten-minute tow is one unit of effort. Trawl data were used to calculate CPUE for BS, WS, BC, and GuM. Nekton collected in Barataria Bay using trawl gear generally targets juveniles (Figures 2.8-2.11).

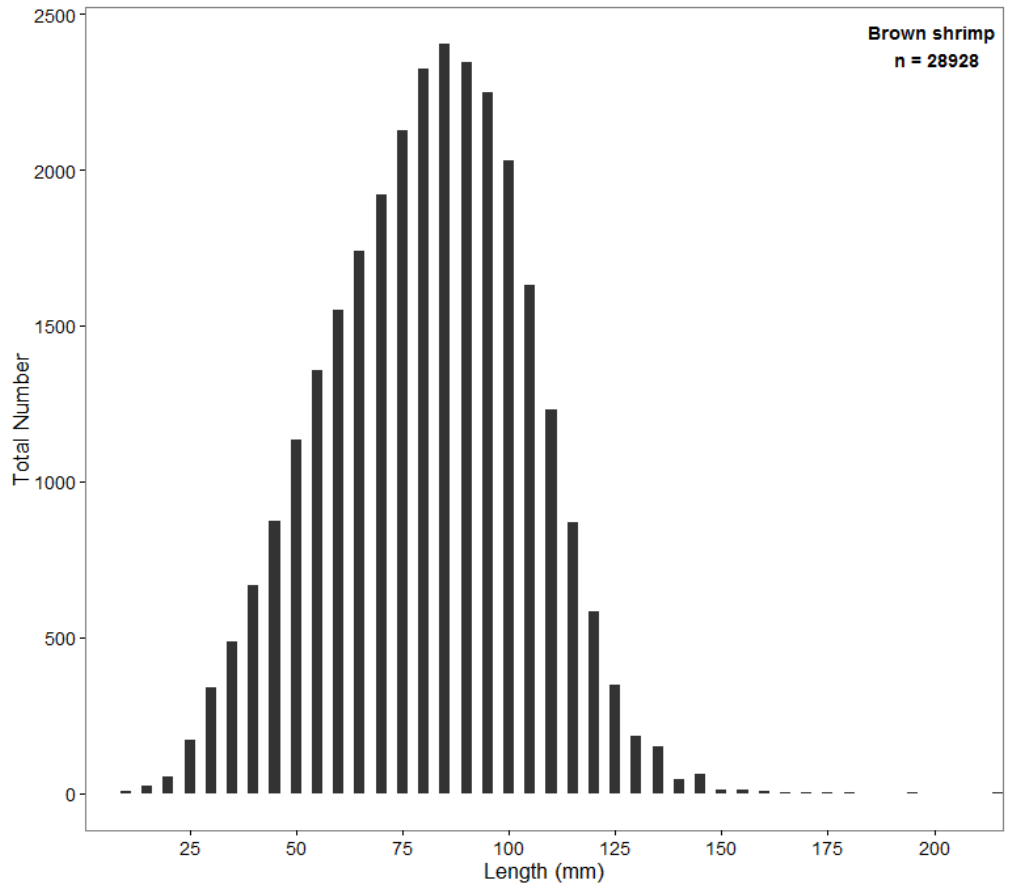

Figure 2.8. Brown shrimp (*Farfantepenaeus aztecus*) length distribution of all organisms measured over the period of record.

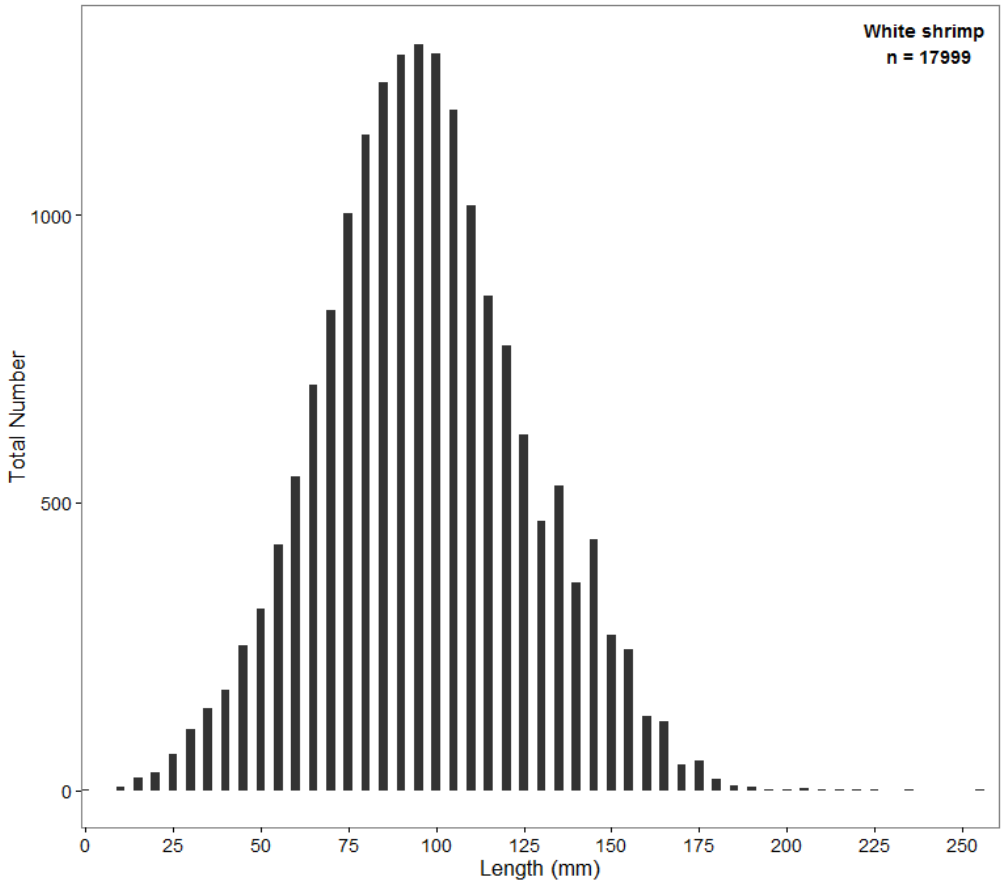

Figure 2.9. White shrimp (*Litopenaeus setiferus*) length distribution of all organisms measured over the period of record.

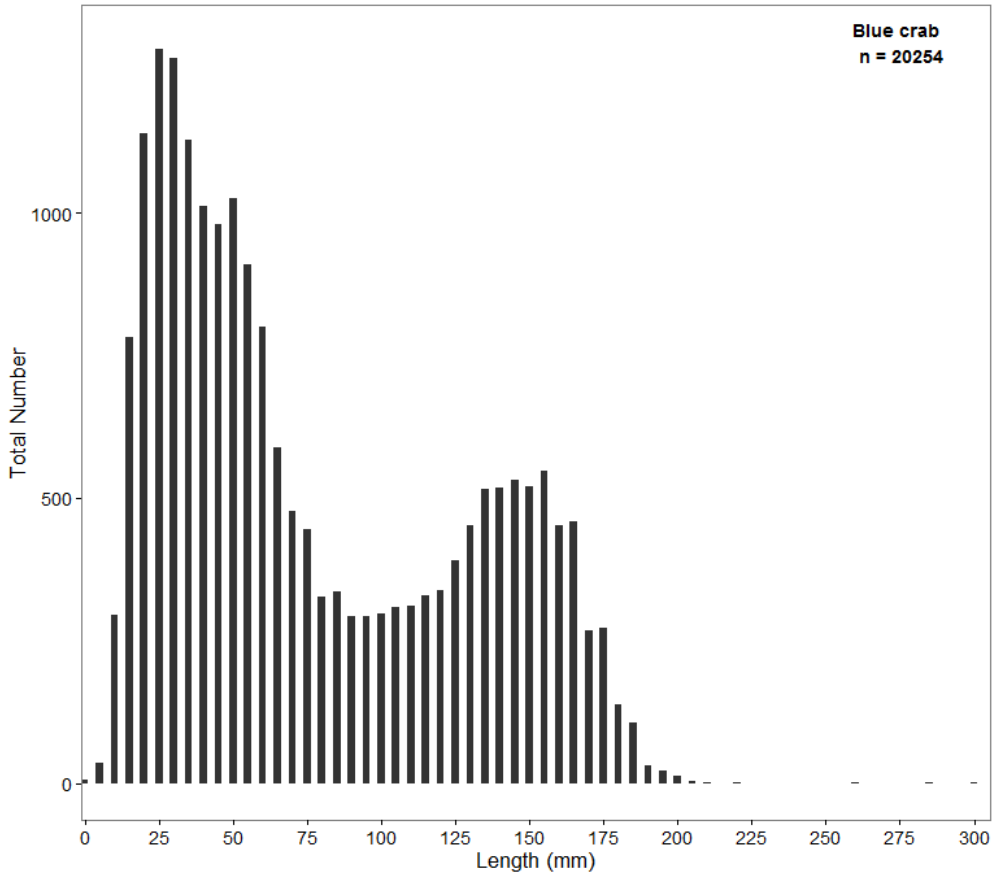

Figure 2.10. Blue crab (*Callinectes sapidus*) length distribution of all organisms measured over the period of record.

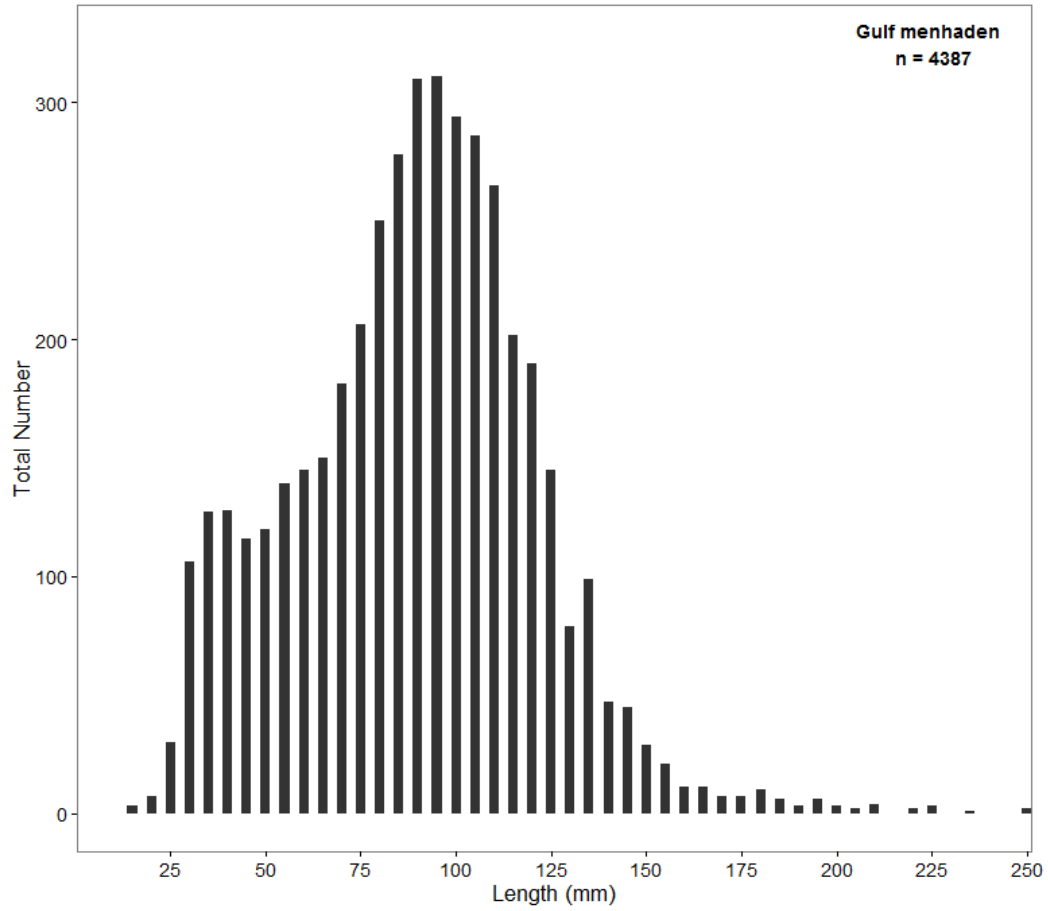

Figure 2.11. Gulf menhaden (*Brevoortia patronus*) length distribution of all organisms measured over the period of record.

Gillnets are 229 m. long and are composed of five-46 m. panels of differing mesh size (2.54 cm., 3.18 cm., 3.8 cm., 4.4 cm., and 5.1 cm.): I combined all mesh sizes in the analyses. LDWF deploys the nets either parallel to the shoreline or in a crescent shape after which a skiff motors around the net in gradually tightening circles. The nets are retrieved, the organisms removed and placed into baskets according to the mesh size in which they were caught. LDWF measures a maximum of 30 organisms per sample (length in mm), with the remaining individuals simply counted and weighed as a group. LDWF collects gillnet samples monthly

from October to March and twice monthly from April to September. Gillnet data were used to calculate CPUE for SStrout and RD. The overall sample size for RD was low in all gear types, but gillnet samples were used because RD fork length was greater in gillnets than in the other gear types. Spotted seatrout were collected using gillnets averaged 300 mm and RD averaged 40 cm in length (Figures 2.12-2.13).

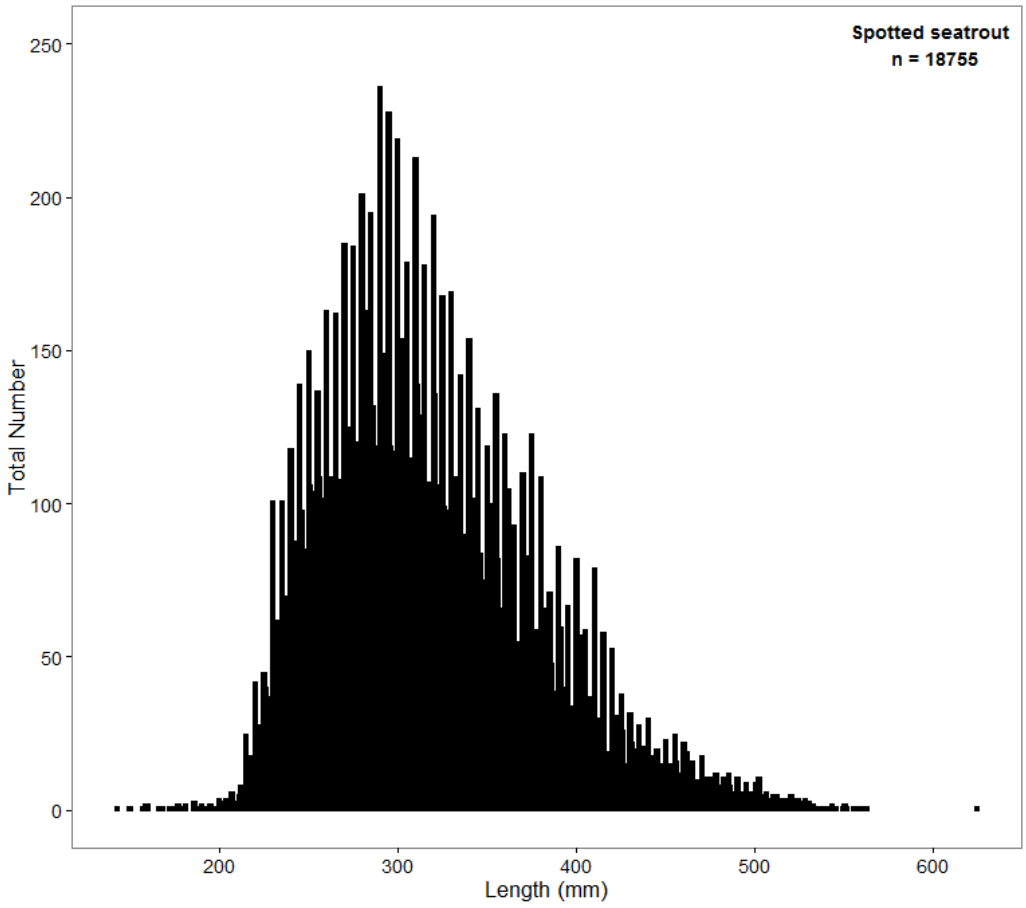

Figure. 2.12. Spotted seatrout (*Cynoscion nebulosus*) length distribution over the period of record.

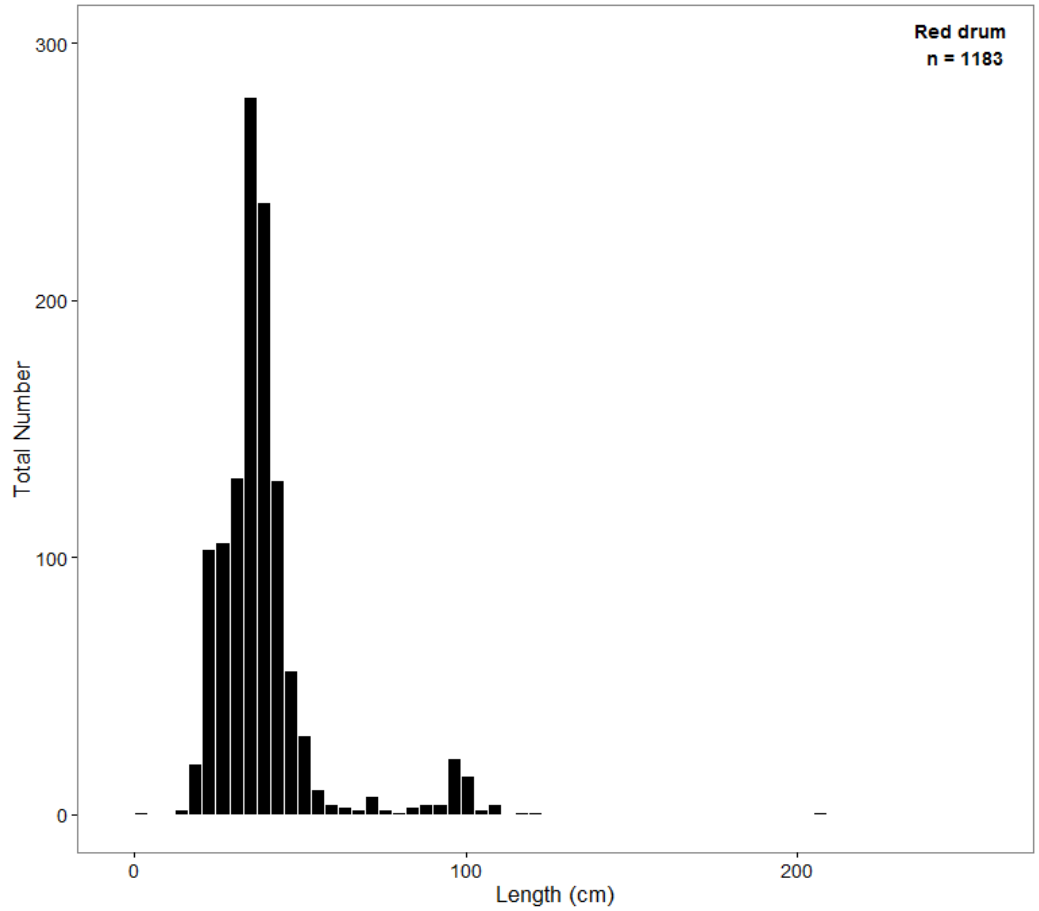

Figure.2.13. Red drum (*Sciaenops ocellatus*) length distribution over the period of record.

Long-term environmental data were obtained from the LDWF survey data; water temperature and salinity are taken at each net set. Readings were taken from the top and bottom of the water column, so I calculated a mean value of each sample and then determined the mean for each year. Estimates of average annual rainfall in Louisiana were obtained from NOAA's National Climatic Data Center (NCDC) (Enloe 2013, Figure 2.14).

I used time series data of linear distance of marsh edge (hereafter, edge) and total marsh area within Barataria Bay. Land-water datasets were obtained from Couvillion (2011) spanning the years 1932-2010 using data sources listed in Table 2.2. I then calculated edge in Barataria

Bay with batch vectorization in ArcGIS® software. To complete the vectorization, all datasets were imported into ArcGIS® as raster images, then clipped to include only the extent of Barataria Basin (using the Intracoastal water way as the northern boundary), and finally each dataset was vectorized into polylines. Once vectorized, edge distance was determined for Barataria Bay by outputting summary statistics within the software. Marsh area was calculated with the Spatial Analyst toolbox within ArcGIS®. I performed a linear interpolation for missing data points with PROC TRANSREG using the SPLINE transformation. This procedure provides a complete time series of predicted measurements for edge and area of marsh (SAS 2013).

Table. 2.2. Data sources used in the creation of the land-water datasets from Couvillion et al. (2011). The number of maps derived from each data source is listed in the last column.

Year	**Data Source**	**No. Maps**
1932	Historical survey data	1
1956	National Wetlands Inventory aerial photography	1
1973-1977	Landsat Multi-Spectral Scanner (MSS)	3
1985-2010	Landsat Thematic Mapper	12

I tested the environmental predictor variables (rainfall, salinity, temperature, marsh area, edge) for normality with normal plots and the Shapiro-Wilks test statistic. I tested for homogeneity of variance and possible time trends in the data by plotting the residuals of each explanatory variable against the response (survey CPUE). The tests indicated that parametric assumptions were met for all variables and in addition, no curvature was observed in the plots.

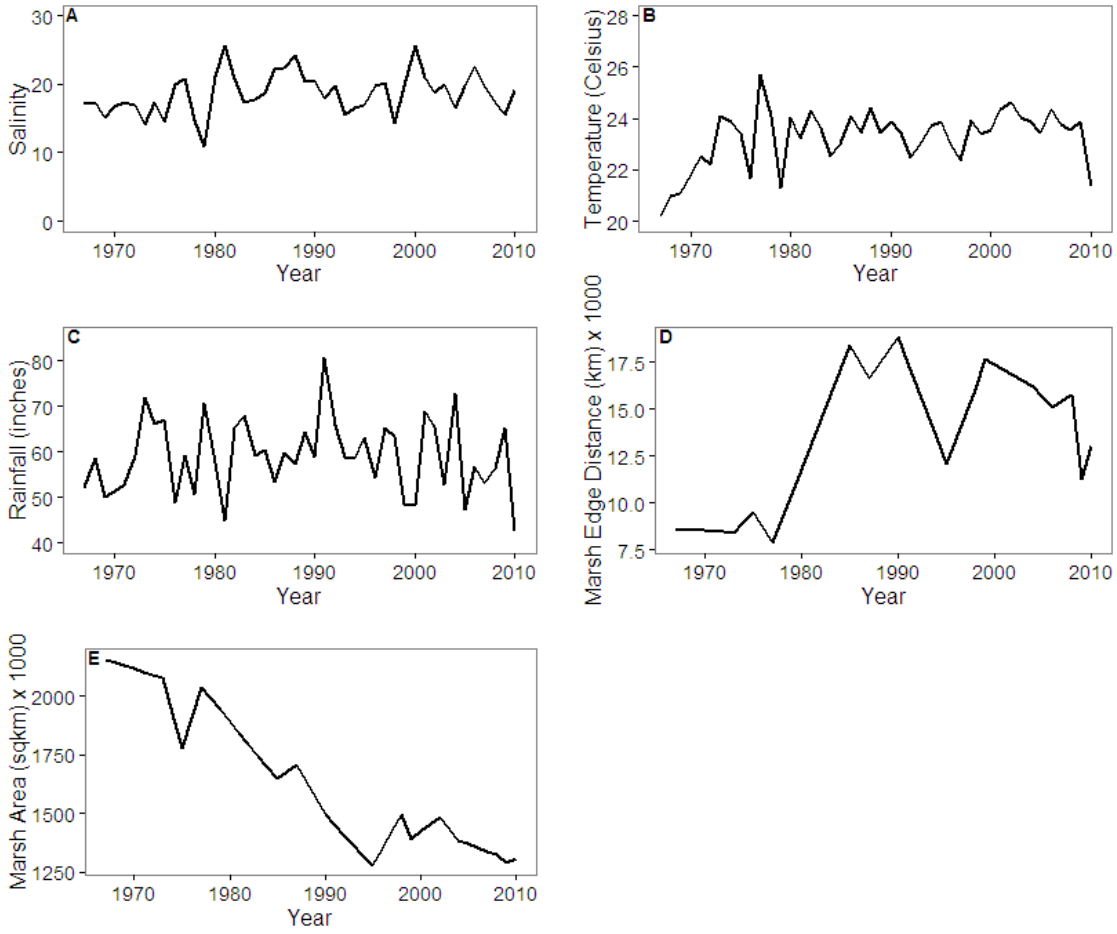

Figure 2.14. Environmental variables for Barataria Bay, LA from 1967-2010. Salinity, and temperature are mean values from Barataria Bay, while rainfall data are actual readings from the NOAA National Climactic Data Center database for all of Louisiana. Marsh edge and marsh area in Barataria Bay were calculated with ArcGIS software from Couvillion (2011).

To develop a descriptive model for each species of interest, I included both fishery independent and fishery dependent catch (landings) data. To determine if the relative abundance (fishery independent data) from the previous year drives the relative abundance (response variable) in the current year, I included a lagged predictor variable (lagged 1, 2, or 3 years depending on life history characteristics of each species). For instance, those species with shorter life histories such as white and brown shrimp were lagged one year, while species that mature more slowly, such as red drum, were lagged three years. Moreover, each of the taxa included in

this study is heavily fished, so to include effects of fishing pressure on the year-to-year variability in the population, landings data were incorporated for each species in the analysis, lagged 1 year. For SStrout and RD, recreational fishery landings data were obtained from NOAA's Marine Recreational Information Program (MRIP) website. Data from MRIP are statewide; therefore fishery independent CPUE data (hereafter survey data) were used to calculate the percentage of SStrout and RD caught in Barataria Bay (over the period of record) when compared with the other major basins in LA. The product of this percentage and the statewide landings provided an estimate of Barataria Bay only landings. Basin specific recreational effort data are not available, so only recreational landings data were used in the analysis (and not recreational CPUE). For BS, WS and BC, basin specific commercial landings and effort data were obtained from LDWF's Office of Fisheries Management for the years 2000 to 2010 and were used to calculate landings CPUE for those three species. After calculating landings CPUE, I determined the percentage of each target species caught in Barataria Bay, and the product of this percentage and total LA commercial landings (derived from NOAA Commercial Landings Statistics) produced estimates of commercial landings specifically from 1967 to 2010 for BS, WS and BC. Effort data were not available for GuM; therefore, the same procedure mentioned above for SS and RD was used to calculate the basin specific landings for GuM.

Annual mean CPUE (by weight) time series were calculated from LDWF survey data (assuming CPUE is proportional to abundance) for each species of interest. Since LDWF only measures 30-50 organisms per sample, I determined the proportion of species at length for each sample. The total catch per sample (when above the 30-50 organism threshold) was then multiplied by the proportional catch-at-size per sample to determine an estimated catch-at-size

for the entire sample. To determine the biomass per sample, catch-at-size were converted to catch-at-weight with a length-weight (L-W) regression, $W_i = aTL_i^b$, where *a* and *b* are the species specific parameters, TL is length in total length, and *i* is each species of interest (Table 2.3). I tested both fishery dependent and survey data for normality, homoscedasticity and curvature and while most of the tests indicated that the error term met the assumption of homoscedasticity, almost all dependent variables did not meet the normality assumption. Therefore, I applied a natural log (ln) transformation to address issues of non-normality and heterscadasticity:

$$lsurveycpue_{jt} = \ln\left(\frac{\sum_{i=1}^{n} catch\_wt_t}{effort_t}\right), \qquad \text{Eq. 2.1}$$

$$lrec\_lbs_{jt} = \ln(landings_{jt}), \qquad \text{Eq. 2.2}$$

$$lcomm\_lbs_{jt} = \ln(landings_{jt}), \qquad \text{Eq. 2.3}$$

$$l\_landings\_cpue_{jt} = \ln\left(\frac{\sum_{i=1}^{n} catch\_wt_t}{effort_t}\right), \qquad \text{Eq. 2.4}$$

where j is the species of interest, t indicates the year, $catch\_wt_t$ is the catch in weight summed over year t , $effort_t$ is the total number of net sets in year t, $landings_{jt}$ is either commercial or recreational landings for each species. All analyses performed in this study utilized the natural log transformed CPUE values and natural log transformed landings for each species when those data were used as response variables, where $lsurveycpue_{jt}$ is the log of LDWF survey data, $lrec\_lbs_{jt}$ is the log of recreational landings data, $lcomm\_lbs_{jt}$ is the log of commercial landings data, and $l\_landings\_cpue_{jt}$ represents commercial landings including effort.

Table 2.3. Weight-length regressions and data sources used to calculate CPUE in biomass. Species indicated with an (*) describe red drum lengths that were converted from TL to FL using a Length-Length regression $FL = (L_{RD} + 2.394)\ x\ (.916)$ as the W-L regression for red drum used FL instead of TL. Species indicated with an (**) illustrates that blue crab were measured using carapace width.

Target Species	Weight-Length Regression	Data Source	Units	
			Weight	Length
Spotted Seatrout	$W_{sst} = 1.13\ x\ 10^{-8}\ TL_{sst}^{3.01}$	(Nieland et al. 2002)	kg	mm
Red Drum*	$W_{rd} = 0.01FL_{bs}^{2.934}$	(Murphy and Taylor 1990)	g	cm
Brown Shrimp	$W_{bs} = 0.006TL_{bs}^{2.938}$	(Fontaine and Neal 1971)	g	cm
White Shrimp	$W_{ws} = 0.003TL_{ws}^{3.247}$	(Fontaine and Neal 1971)	g	cm
Blue Crab**	$W_{bc} = 0.008CW_{bc}^{2.45}$	(West 2013)	g	mm
Gulf Menhaden	$W_{gm} = 0.008TL_{gm}^{3.02}$	(De Mutsert 2010)	g	cm

Statistical Analysis 2.2.3.

I regressed survey data on both landings data (all species) and landings CPUE data (for brown shrimp, white shrimp, and blue crab) to test the null hypothesis that landings data are an appropriate index of relative abundance for each species. Time series plots were created to visually inspect historical trends using survey, landings, and landings CPUE data.

To test the null hypothesis that species abundance varies randomly through time, a multiple linear regression analyses was conducted for each species of interest. First, to identify possible cases of multicolinearity, a correlation matrix was created to look at pairwise relationships. To investigate more than just pairwise multicollinearity, I regressed survey data on all environmental variables (a "full" model) to determine if different combinations of explanatory variables correlate with other combinations. Variance Inflation Factor (VIF) values

were investigated to determine if multicollinearity was having a negative influence on least square estimates. Since time-series data were used in this analysis, a Durbin-Watson test for autocorrelation was applied. The predictor variables in this analysis cover a wide range of values, deeming it necessary to re-parameterize the model to account for these large disparities. To address this issue, I "centered" and "scaled" the explanatory variables by subtracting the overall mean from each observation and assigning a standard deviation of one. Centering can also address issues of multicollinearity that remain in the model (Kutner et al. 2005).

Model selection for this analysis was a three-fold process. First, an "all possible regression" approach was used by considering all possible predictor combinations (which is $2^{(p-1)}$, where p is equal to the number of predictors in the full model) and identifying the top three "best" subset models according to the Akaike's Information Criterion (AIC), which penalizes models for having a large number of predictors (Kutner et al. 2005). Next, a backward elimination multiple linear regression was used to determine a single "best" regression model. This method begins with all variables of interest in the model and iteratively removes variables that do not meet a selection criteria, here $\alpha = 0.05$. I also tested quadratic trends of predictor variables in both model selection processes to allow the models to fit a curve, if necessary. Lastly, I tested a number of different models using a priori ecological information about each species. For instance, temperature and salinity are known to drive SStrout abundance and distribution. Therefore, I constructed a number of models that included those variables in different combinations with other previously tested significant predictors. I evaluated the final subset of models by comparing Adjusted R^2 and AIC values to determine the "best" model for each species of interest. The multiple regression analyses were fitted with PROC REG in SAS v. 9.3 (SAS 2013).

2.3. Results

2.3.1. Summary statistics for survey data

Survey samples were 50% (or more) effective at catching species of interest for four of the organisms considered (SStrout, BS, WS, BC), while RD and GuM were far less effective (Table 2.4).

Table 2.4. Summary statistics for the LDWF fishery survey data that were used in CPUE calculations.

Target Species	Gear Type	Years Sampled	Total Catch	Total Pounds	No. of Net Sets	No. Pos Net Sets	% Positive
Spotted Seatrout	Gillnet	24 1986-2010	18169	16854.02	4630	2286	49%
Red Drum	Gillnet	24 1986-2010	1181	2626.54	4630	397	9%
Brown Shrimp	Trawl	43 1967-2010	228779	3519.4	9044	5111	57%
White Shrimp	Trawl	43 1967-2010	75951	1721.97	9044	4269	47%
Blue Crab	Trawl	43 1967-2010	38852	3866.46	9044	4751	53%
Gulf Menhaden	Trawl	43 1967-2010	30179	635.88	9044	1511	17%

2.3.2. Survey and Landings Comparisons

Three different data sources were considered to determine if survey data showed similar trends when regressed on landings and landings CPUE data. For all but one species (WS), landings data and landings CPUE data appear unrelated to survey data. While five of the nine regression analyses showed statistical significance, the R^2 values were extremely low except for survey and landings CPUE data for WS (Table 2.5).

Table 2.5. LDWF survey CPUE data regressed against landings and landings CPUE data for each species in Barataria Bay, LA. Species indicated with an (*) shows a statistically significant relationship between the two sets of data ($\alpha < .05$).

Species	Regression Data	Param. Estimate	F	p	R^2
Spotted seatrout*	survey & landings	0.22	6.12	0.02	0.18
Red drum	survey & landings	0.15	2.35	0.14	0.05
Brown shrimp*	survey & landings	0.14	5.59	0.02	0.10
Brown shrimp	survey & landCPUE	0.13	0.18	0.68	0.09
White shrimp	survey & landings	0.07	2.8	0.10	0.04
White shrimp*	survey & landCPUE	0.24	8.85	0.02	0.44
Blue crab*	survey & landings	-0.23	4.54	0.04	0.08
Blue crab	survey &landCPUE	0.08	0.90	0.37	0.01
Gulf menhaden*	survey & landings	0.01	0.03	0.87	0.02

When long-term patterns of survey data are compared with landings data, a few notable trends emerge. Survey data indicate that relative abundance of SStrout is decreasing while landings data show a clear increasing trend in harvest (Figure 2.15). Survey data indicate that RD abundance is stable in Barataria Bay while landings are continuing to increase (Figure 2.16). Brown shrimp survey data showed no trend over the period of record, while landings CPUE data showed a slightly increasing trend (Figure 2.17A).

Though the survey data showed a strong decreasing trend for BC, more recent years indicate there has been a general upturn in blue crab relative abundance, which is also reflected in landings CPUE data (Figure 2.19B). Landings data also fluctuate through time, but generally shows a stable or increasing trend (Figure 2.19B).

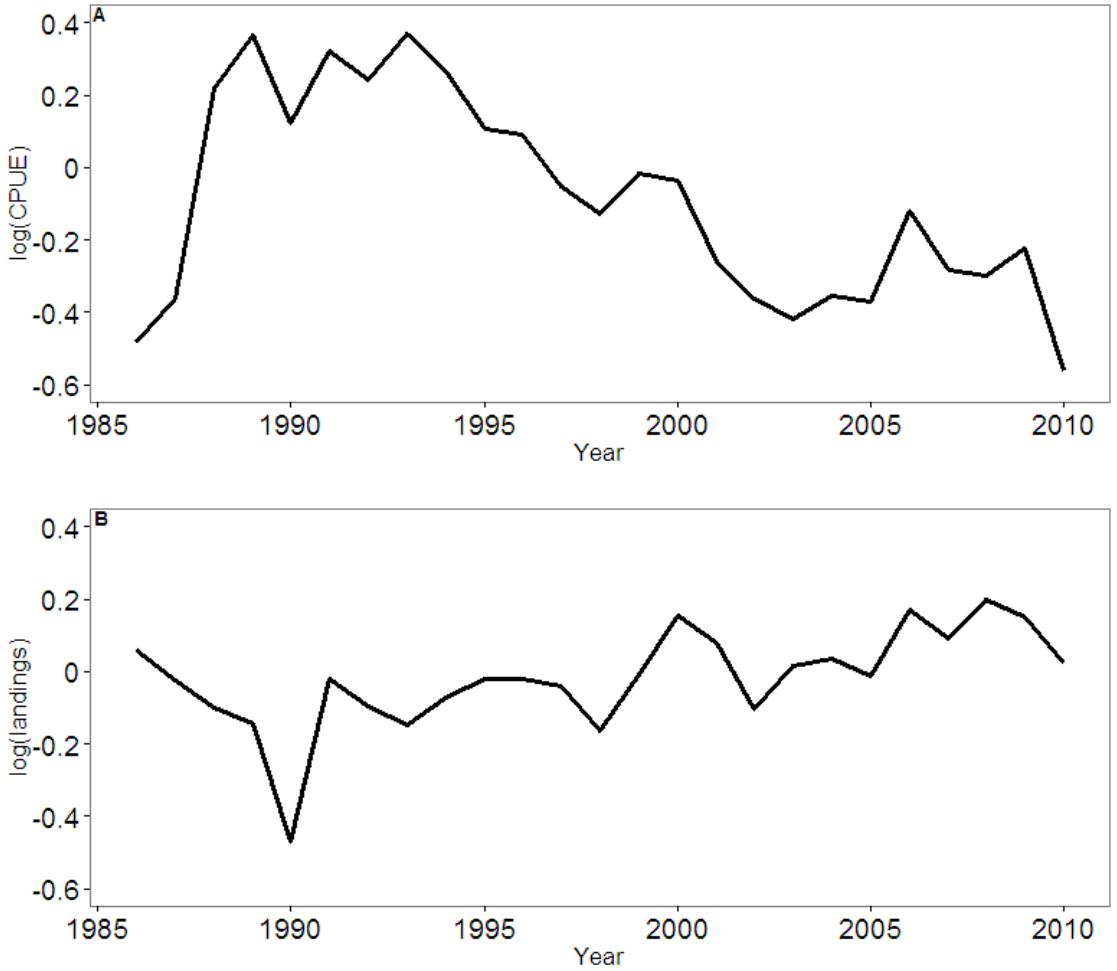

Figure 2.15. Time series plots of spotted seatrout (*Cynoscion nebulosus*) CPUE survey data (A) and spotted seatrout recreational landings data (B).

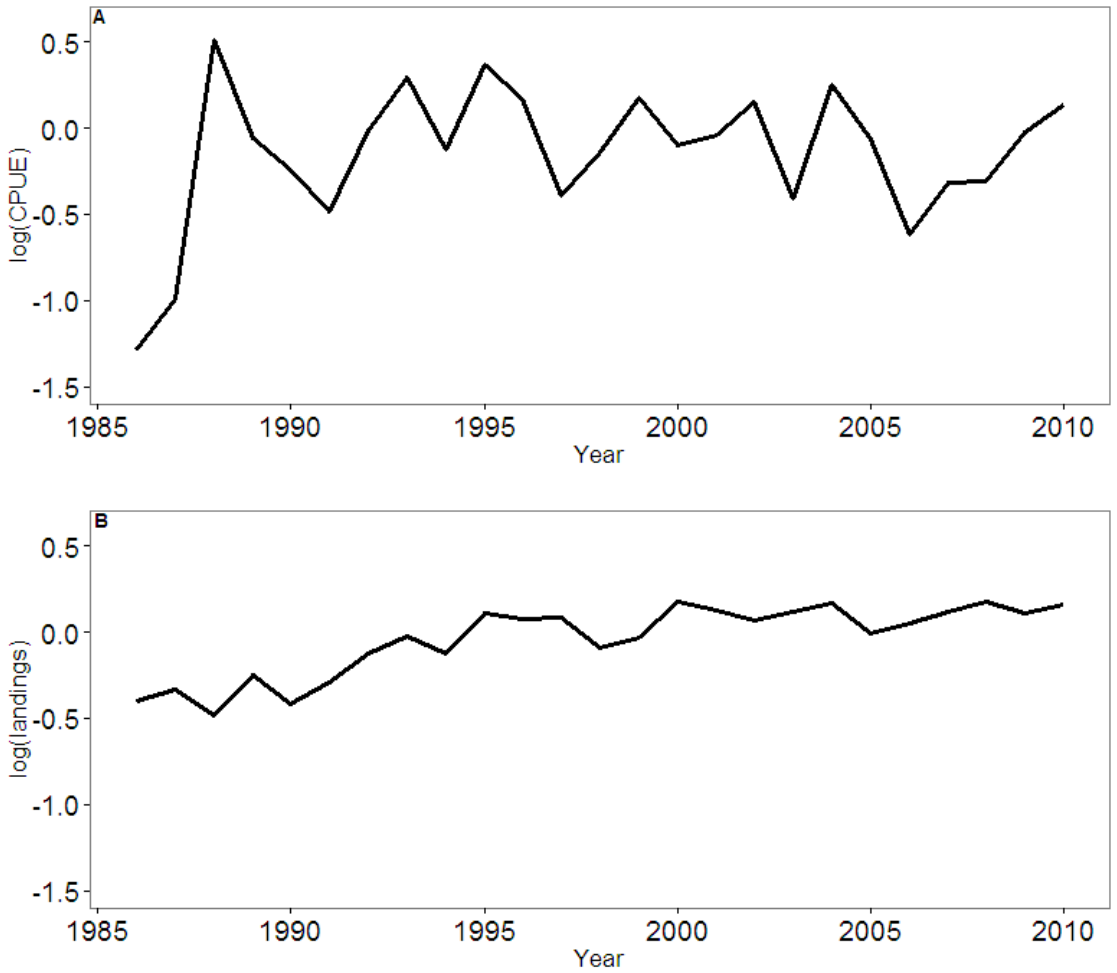

Figure 2.16. Time series plots of red drum (*Sciaenops ocellatus*,) survey CPUE data (A) and red drum recreational landings data (B).

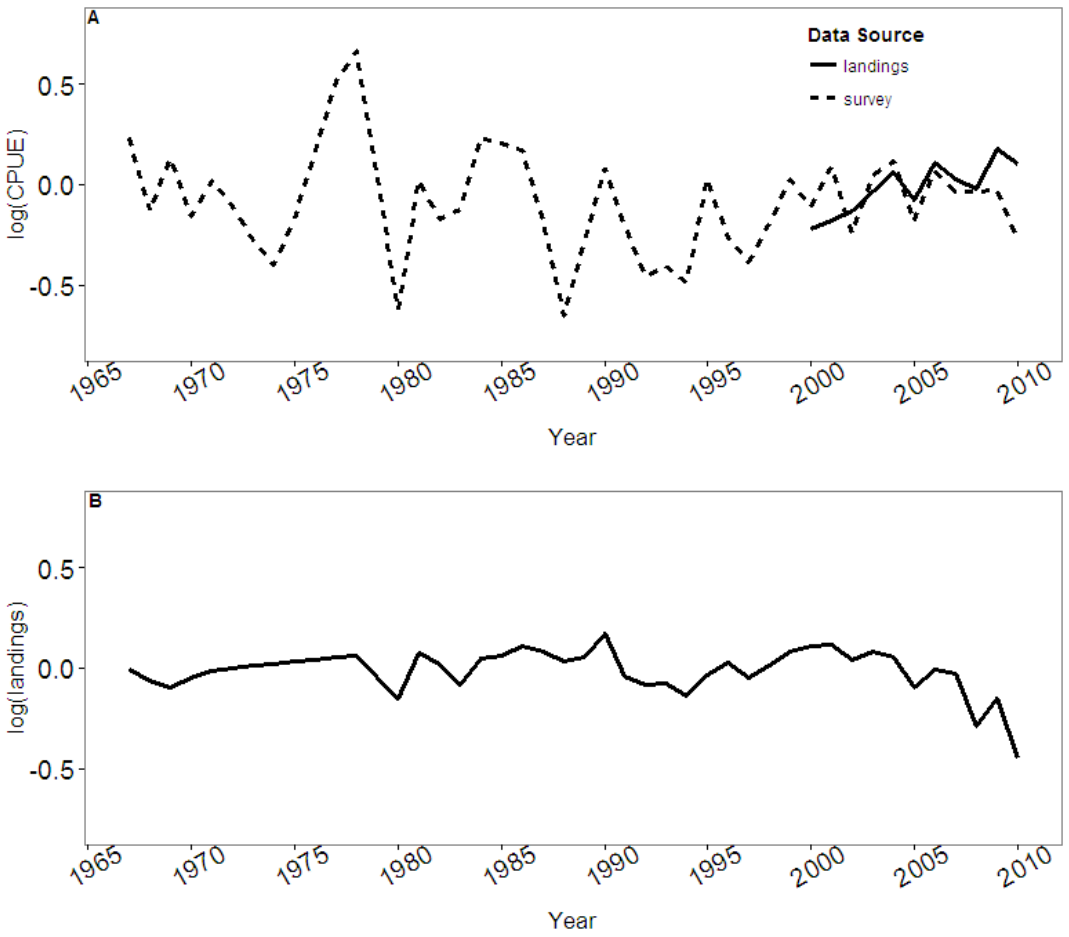

Figure 2.17. Time series plots of brown shrimp (*Farfantepenaeus aztecus*) survey CPUE data (dashed line) and brown shrimp commercial landings CPUE data (solid line) (A) and brown shrimp commercial landings data (B).

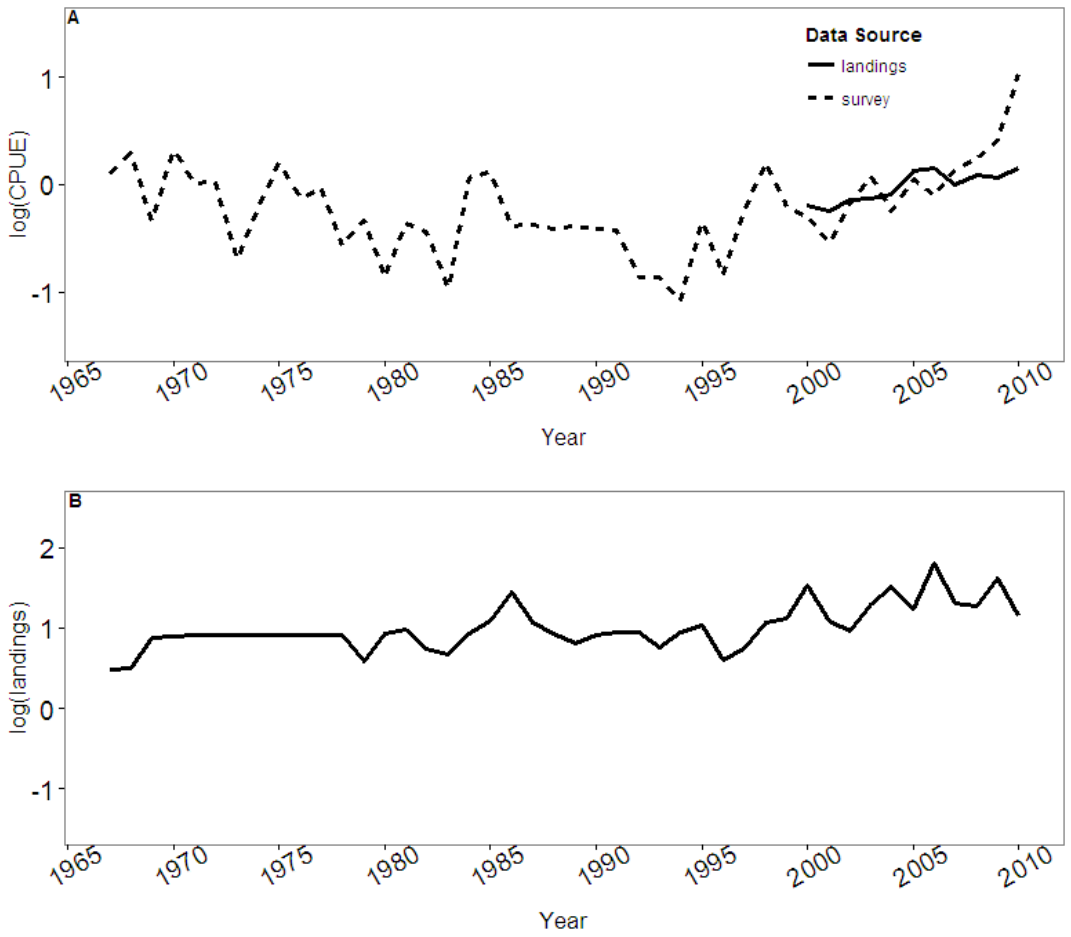

Figure 2.18. Time series plots of white shrimp (*Litopenaeus setiferus*) survey CPUE data (dashed line) and white shrimp landings CPUE data (solid line) (A) and white shrimp commercial landings data (B).

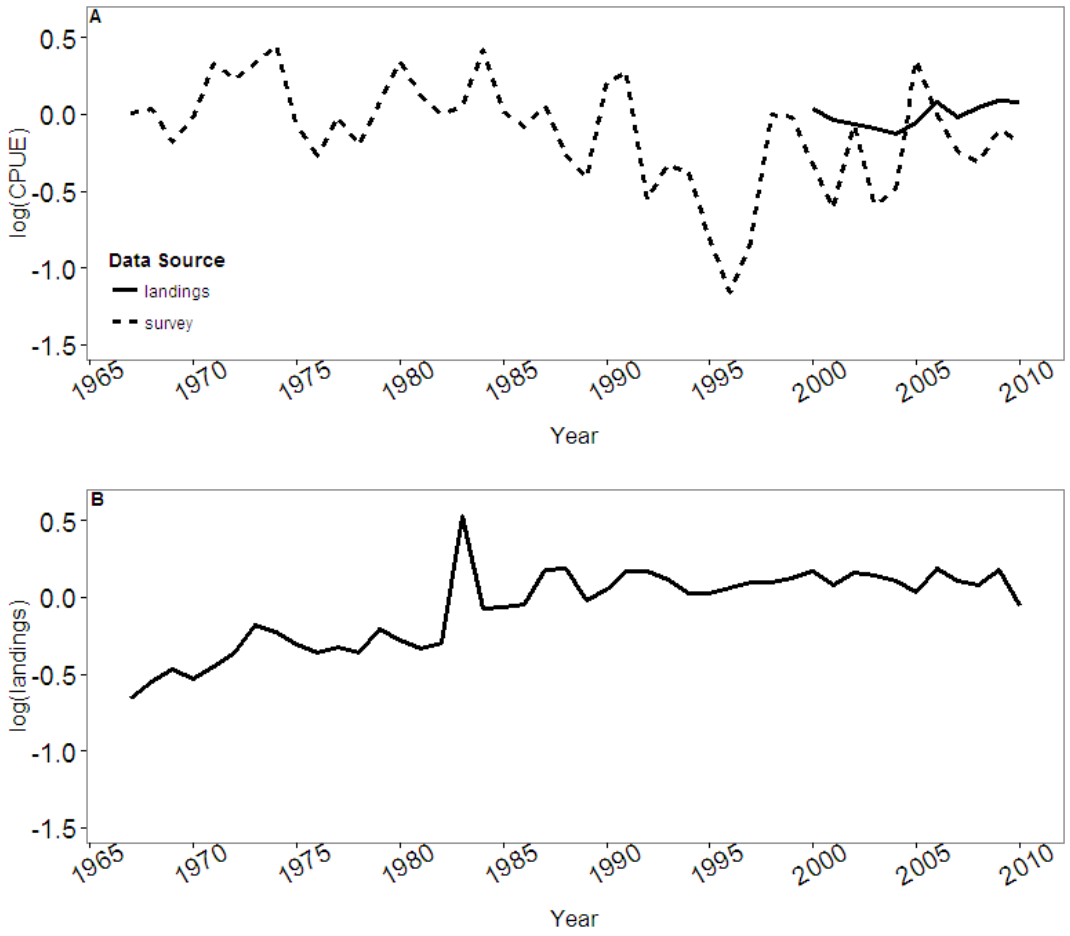

Figure 2.19. Time series plots of blue crab (*Callinectes sapidus*) survey CPUE data (dashed line) and blue crab commercial landings CPUE data (solid line) and blue crab commercial landings data (B).

Gulf menhaden survey data indicated variable relative abundance over time, with an overall decreasing trend, while the landings data for GuM did not show any specific pattern of change over time (Figure 2.20).

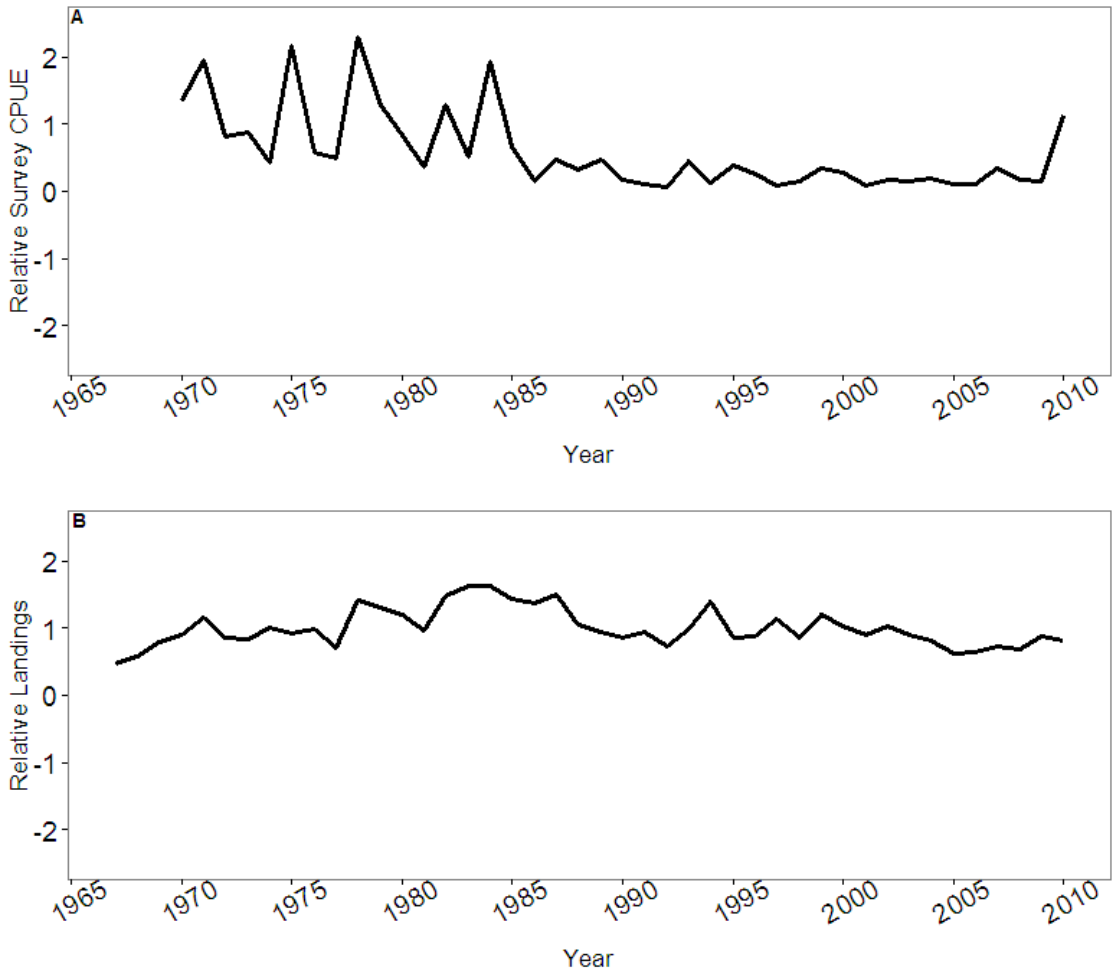

Figure 2.20. Time series plots of Gulf menhaden (*Brevoortia patronus*) survey CPUE data (A) and Gulf menhaden commercial landings data (B).

2.3.3. Model development

Results of the Pearson-Product Moment correlation analysis on the standardized environmental variables indicated negative correlation between the area of marsh and the linear distance of marsh edge (Tables 2.6 and 2.7). Regression of the "full" model indicated that when all explanatory variables were included in the analysis for each species of interest, there was indeed an issue of multicollinearity ($\overline{VIF} > 3$). However, no remedial measures were taken (except the previously mentioned centering and scaling of predictor variables) because variable selection procedures produced reduced models that were free of multicolinearity. The "full"

model was also used to determine if autocorrelation was present in the time series data. The Durbin-Watson test indicated that there were no issues of serial correlations with any time series data.

Table 2.6. Pearson correlation coefficients for all standardized environmental variables (trawl data) considered in the regression analyses. The variables include salinity (sal), water temperature (temp), rainfall (rain), linear distance of marsh edge (edge), and area of marsh (marsh). Variables ending in a "2" indicate the quadratic form of each predictor.

	sal	**temp**	**rain**	**edge**	**marsh**	**sal2**	**temp2**	**rain2**	**Edge2**	**Marsh2**
sal	**1.00**	0.31	-0.38	0.46	-0.24	0.08	-0.16	-0.05	-0.17	-0.41
temp	0.31	**1.00**	0.27	0.38	-0.37	0.00	-0.51	-0.12	-0.20	-0.50
rain	-0.38	0.27	**1.00**	0.13	-0.07	-0.05	-0.18	0.18	-0.03	-0.24
edge	0.46	0.38	0.13	**1.00**	-0.72	-0.05	-0.52	0.05	-0.44	-0.76
marsh	-0.24	-0.37	-0.07	-0.72	**1.00**	0.15	0.52	-0.09	0.71	0.48
sal2	0.08	0.00	-0.05	-0.05	0.15	**1.00**	0.00	0.16	-0.08	-0.14
temp2	-0.16	-0.51	-0.18	-0.52	0.52	0.00	**1.00**	-0.02	0.43	0.60
rain2	-0.05	-0.12	0.18	0.05	-0.09	0.16	-0.02	**1.00**	-0.06	-0.03
Edge2	-0.17	-0.20	-0.03	-0.44	0.71	-0.08	0.43	-0.06	**1.00**	0.41
Marsh2	-0.41	-0.50	-0.24	-0.76	0.48	-0.14	0.60	-0.03	0.41	**1.00**

The most parsimonious models were selected by choosing the models with the smallest AIC value. For all but one species (GuM), the AIC values between the "best" and the "worst" were not considerably disparate (Table 2.8). Of the six species considered in this analysis, three species were found in which habitat (marsh area or edge) was an important factor in explaining variations in abundance. Statistically significant models were also found for WS and SStrout. Models tested for RD failed to explain a significant portion of the variability in their abundance.

Table 2.7. Pearson correlation coefficients for all standardized environmental variables (gillnet data) considered in the regression analyses. The variables include salinity (sal), water temperature (temp), rainfall (rain), linear distance of marsh edge (edge), and area of marsh (marsh). Variables ending in a "2" indicate the quadratic form of each predictor.

	Sal	**temp**	**rain**	**edge**	**marsh**	**sal2**	**temp2**	**rain2**	**edge2**	**marsh2**
sal	**1.00**	-0.16	-0.02	0.24	0.44	-0.06	0.09	-0.20	0.07	0.35
temp	-0.16	**1.00**	0.07	-0.21	-0.38	-0.05	-0.41	-0.05	0.11	-0.33
rain	-0.02	0.07	**1.00**	0.10	0.12	-0.06	-0.17	0.21	0.04	-0.12
edge	0.24	-0.21	0.10	**1.00**	0.68	0.10	-0.36	0.10	-0.54	0.05
marsh	0.44	-0.38	0.12	0.68	**1.00**	0.11	-0.09	-0.14	-0.30	0.66
sal2	-0.06	-0.05	-0.06	0.10	0.11	**1.00**	-0.28	-0.14	0.17	0.15
temp2	0.09	-0.41	-0.17	-0.36	-0.09	-0.28	**1.00**	-0.11	0.16	0.26
rain2	-0.20	-0.05	0.21	0.10	-0.14	-0.14	-0.11	**1.00**	-0.04	-0.26
edge2	0.07	0.11	0.04	-0.54	-0.30	0.17	0.16	-0.04	**1.00**	0.11
marsh2	0.35	-0.33	-0.12	0.05	0.66	0.15	0.26	-0.26	0.11	**1.00**

Table 2.8. Final regression models determined by the multiple regression analysis. The variables include response survey CPUE lagged 1 year ($lcpue_{t-1}$), salinity (sal), water temperature (temp), rainfall (rain), linear distance of marsh edge (edge), and area of marsh (marsh), and recreation landings lagged 1 year (rec_lbs_{t-1}). Variables that are squared indicated a quadratic predictor. All environmental variables were standardized to a mean of zero and a standard deviation of 1.

Final Models	F	p	R^2	$CV\%$
$lcpue_{sst} = lcpue_{t-1} + sal - rec\_lbs_{t-1}$	20.86	<0.0001	0.68	4.93
$lcpue_{RD} = temp + temp^2$	2.76	0.09	0.14	10.39
$lcpue_{BS} = lcpue_{t-1} + edge + edge^2$	3.76	0.02	0.33	11.01
$lcpue_{WS} = lcpue_{t-1}$	11.44	0.002	0.20	28.98
$lcpue_{BC} = lcpue_{t-1} + marsh$	11.48	0.0001	0.33	13.29
$lcpue_{GM} = temp + marsh$	33.22	<0.0001	0.61	26.57

2.4. Discussion

This study found that using landings data alone does not accurately represent historical abundance when survey data are compared with landings data. While landings data might suggest that fishery production is remaining strong for SStrout and RD over the long-term (Figure 2.4 and 2.5), this conclusion poses a problem when landings and survey data are considered concurrently. Effort data (among other potential sources of bias) are not considered in landings data extracted from the NOAA MRIP (recreational landings source) and using these data alone (without effort) to assess overall patterns of abundance would be uncertain. After observing the increase of effort in the Louisiana recreational sector, an increase in landings for SStrout and RD is not surprising, since recreational effort has shown a steady increase since 1990 (Figure 2.21). Regression of survey data on landings data for both SStrout and RD suggests that there is not a strong relationship between survey and landings data (Table 2.4; Figures 2.15-2.16). The multiple regression analysis for SStrout showed that variation in abundance for this species is dependent on salinity, in addition to the previous year's catch and the previous year's relative abundance (Table 2.8). In light of that information, care should be taken in terms of management decisions for this species as it appears annual variations are driven by both environmental factors and fishing related pressures. On the other hand, this study was not able to find a suite of explanatory variables that significantly explained the variability in RD abundance over the period of record (Table 2.8). An obvious explanation for the lack of a significant model is that the variables that drive RD abundance were not included in the analyses. Another explanation could be that the sample size for RD was low in Barataria Bay. Nevertheless, considering the trends in recreational landings (increasing) along with the lack empirical information that could help explain the variability in RD abundance, I highly

encourage managers to use a precautionary approach when setting creel limits and other regulations for this species (Darcy and Matlock 1999).

Figure 2.21. Recreational effort for Louisiana from 1986-2012 (NOAA-MRIP 2013).

Caution should also be used in evaluating the outcome of the recreational species in this study because of the uncertainty introduced when extrapolating the data to Barataria Bay. Still, this information does draw attention to the importance of using survey data to analyze long-term trends in important fisheries. This is not to say that landings data are unimportant in the successful management of fish and shrimp species. It is understood that landings data are an essential component in developing stock assessments. Recently, LDWF has vastly improved its stock assessment methodologies by including indices of abundance drawn from the survey data (West et al. 2011). Other stock assessments for important species are being updated in the same manner, but it will take considerable time to complete these analyses. In the meantime, managers should consider carrying out cursory analyses, similar to the approach in this study, until the new stock assessments are available as continued increases in effort and increases in creel limits could negatively impact the standing stock prior to the stock assessments being updated.

Brown shrimp landings have been steadily decreasing over the last 15 years, but when effort data are used to calculate landings CPUE from 2000 to 2010, the trend actually increases. The trend between BS landings CPUE and BS survey CPUE is similar at times, but diverges in the most recent years reported. The decrease in brown shrimp landings can be explained by an overall decrease in effort over the past 10-15 years (Nance 2011). A number of factors have led to the decrease of effort in the commercial shrimping industry, including a decrease in infrastructure, declining shrimp prices due to competition from Asian shrimp markets, increased gas prices, and the 2010 BP oil spill. These results indicate that using landings data alone to describe production in the brown shrimp fishery would lead to conclusions that may not accurately describe relative densities of this species in Barataria Bay. Moreover, even when landings CPUE data are compared to survey data trends, there is no significant relationship between the two indices ($p = 0.68, R^2 = -0.09$). In addition to the possibility that landings data alone may cause misrepresentative conclusions about biomass dynamics, basin differences are also an important consideration in this dynamic ecosystem. Louisiana state fisheries managers and stock assessment scientists are already aware of the possible regional differences in shrimp production as they have separated the LA coast into 3 different management regions from east to west. The results found herein are meant to inform future investigations for analyzing overall fisheries data from LA (other than stock assessments). I contend that aggregating species and basins in LA will lead to conclusions that may not accurately describe system dynamics.

While WS survey data and landings CPUE track rather well, survey data actually show a stronger increase in abundance. Further bolstering this relationship, regression analysis of the survey date and landings CPUE data of WS showed the strongest relationship out of all taxa

considered ($R^2 = 0.44$). It is not clear why all WS patterns of abundance seem to agree, while BS abundance estimates do not. However, the results from the multiple regression analysis might shed some light on these differences. The regression model for BS shows that the variability in their abundance can be explained in part by a quadratic relationship with the linear distance of marsh edge (Table 2.8). On the other hand, the model selection procedure failed to attribute any habitat factor to the variability in WS abundance. These results potentially suggest that WS are particularly robust in the face of marsh area and edge loss. Previous studies however, have well documented the importance of both internal marsh habitat and edge as significant factors in WS production (Rozas and Reed 1993, Minello and Rozas 2002, Rozas et al. 2007). The results of this study from one basin in LA are not conclusive enough to state that WS are less dependent on marsh habitat than once thought. Nonetheless, if there is any degree certainty behind these findings, the ability for WS abundance to remain stable over time concurrent with extreme losses of marsh becomes much more believable. Therefore, considering the loss of marsh area and edge that is currently taking place in Barataria Bay, it would appear that BS may be more slightly susceptible to these changes in habitat than white shrimp. Although, the parabolic relationship found with BS and edge is not easily explained in ecological terms so the previous statement is made cautiously.

The model selection procedure found marsh area to be of particular importance in explaining variability in BC abundance. Given previous studies relating BC density with marsh habitat, along with the increase in the landings data over the same time period, one could speculate that both factors contributed to the decline in overall abundance of BC seen in the survey data (Zimmerman et al. 2000). The opposing trends for survey data and commercial landings data (without effort) purports the notion that using landings data as indicator of relative

abundance in this instance clearly would present an ambiguous trend in the ecosystem. However, with the creation of the Louisiana Trip Ticket program in 2000, effort data is now reported with BC catch, allowing for the calculation of landings CPUE for this species. Landings CPUE for BC does in fact track rather well over the past 10 years. So, in this instance, if landings data were used for any analysis, rather than survey data or landings CPUE data, results may not reflect actual ecosystem dynamics. Here again, with the lack of state or even basin specific stock assessments for BC tuned to survey data, it will be important for managers to consider using survey data or landings CPUE data at the very least to help inform management of blue crab in Louisiana.

Gulf menhaden survey data indicates a strong decrease in relative abundance over time (Figure 2.20). Other studies have shown that GuM landings have actually increased over the past 30 or 40 years, but those data are aggregated over all of LA (Chesney et al. 2000). This study shows that landings in Barataria Bay have remained generally stable over the period of record. If GuM relative abundance in Barataria Bay is decreasing, while landings are remaining the same, fishing effort must to be increasing. However, effort data for the GuM fishery is confidential, so this investigator has no manner by which to substantiate this hypothesis. If it is indeed the case, then the continued increases in effort could have catastrophic effects on this fishery. The life history of GuM does lend itself to resiliency, however the predictive model chosen for GuM in this study indicated a relationship between marsh area and menhaden density (in addition to water temperature). While a regression relationship does not imply causality, the area of marsh was chosen as a significant factor to explain historical variability for this species (Table 2.8). The loss of marsh area in Barataria Bay, therefore, could be another influencing factor in the overall decline in GuM abundance. Menhaden collected in this study were

primarily juveniles, known to have a much stronger affinity for marsh habitat than the adults, which helps to corroborate the results found in this study (Baltz et al. 1993). Stock size has no observable influence on recruitment in GuM and any relationship could be undetectable due to unaccounted for environmental factors (Vaughan et al. 2007). It follows then that the findings in this study, which potentially relate GuM abundance to area of marsh and fishing pressure, could be influencing GuM abundance over the long term. Expanding these methods to other basins in addition to having access to the GuM fishing effort, future analyses could confirm (or refute) the trends found in this dissertation.

The purpose of this study was to highlight the varying trends between survey, landings, and landings CPUE while examining external drivers in the system that might affect long term abundance. Previous studies justify the use of landings data for various analyses because species with shorter life spans, such as BS, WS, and GM, will complete their life cycle in approximately a year. This study suggests that in Barataria Bay, LA, using landings data alone to describe relative abundance would misrepresent abundance estimates, a fact that could be detrimental to both the fishermen and the ecosystem. While it is clear that fishery independent data are not available in many systems, that situation is not the case in Louisiana. Yet, studies in Louisiana still cite relationships of fish production using landings data with factors such as habitat (Browder et al. 1989, Pauly and Ingles 1999, O'Connor and Matlock 2005). While stock assessments in Louisiana do incorporate survey data in their analyses, this study hopes to encourage future researchers outside of stock assessment science to take advantage of the fishery independent database provided by LDWF. In addition, continuing to explore the environmental factors highlighted in the regression analyses herein, which affect long term abundance of estuarine nekton, is of great importance due to the rapidly changing ecosystem. Moving forward,

utilizing fishery independent monitoring data, along with emerging data for wetland loss in each individual coastal basin, will help to further the understanding of both spatial and temporal dynamics of nekton species.

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CHAPTER 3.
HISTORICAL ANALYSIS OF HABITAT LOSS AND ESTUARINE NEKTON ABUNDANCE IN BARATARIA BAY, LOUISIANA, USA

3.1. Introduction

For almost 40 years, scientists have hypothesized that Louisiana fisheries could be in danger of collapse due to considerable loss of marsh, an aspect that is thought to play an important role in fisheries recruitment and growth (Turner 1977, Browder et al. 1989, Nance et al. 1989, Chambers 1992, Zimmerman et al. 2002, Haas et al. 2004, Valiela et al. 2004, Cowan et al. 2008). Although marsh loss is high ~80 km^2 d^{-1}, fishery landings in Louisiana do not appear to be negatively impacted, a response not fully explained by decreases in effort (Nichols 1984, Chesney et al. 2000, Zimmerman et al. 2002). Previous studies have correlated fisheries landings with the extent of marsh area (Turner 1977, Browder et al. 1989, Pauly and Ingles 1999). While landings data are the most available form of fisheries data, it is widely held that landings data are inherently biased when used as indices of abundance and not standardized. Results from Chapter 2 indicate that relative abundance data of most nekton species analyzed fails to show similar patterns of abundance when compared to landings data. Studies that have correlated landings with the area of intertidal wetlands may be confounded by other factors such as changes in effort, improved technology and economic drivers (Lee 2004). To address some of these limitations, investigating marsh loss with the use of long-term fishery independent data may provide insights that using landings data would obfuscate.

The leveeing of the Mississippi River (MR) interrupted the natural deltaic cycle in LA, a cycle which has been occurring for thousands of years (Coleman 1988). By its very nature, delta lobe formation and abandonment have long presented a changing habitat to those species which

occupy the estuaries. Therefore, the resilience seen by many estuarine nekton, may simply be a result of those species evolving over time to deal with a constantly changing system. However, direct empirical evidence (e.g. increased growth rates or increased recruitment) supporting this hypothesis has been elusive and studies testing the relationship between nekton and marsh habitat have mainly been conducted in mesocosms and not in nature.

Much of the literature regarding marsh loss and fish abundance focuses on what I call the edge effect, first purported by Browder et al. (1985). The edge effect is the process by which the marsh degrades and the amount of marsh edge interface (hereafter edge) initially increases. Increased edge potentially increases access to the marsh by estuarine nekton. This increased access to the marsh theoretically benefits organisms that may utilize the edge for foraging opportunities and refuge during the initial stages of marsh degradation. Browder et al. (1989) suggested that once an estuary becomes more than 50% open water (at which point edge begins a steep decline), fisheries could be negatively impacted as organisms will have less opportunity to access the marsh. Previous studies have shown that densities of nekton are higher at the edge, and that densities decrease precipitously as one moves away from the edge (Minello et al. 1994, Zimmerman et al. 2002, Rozas et al. 2007). Yet, evidence is beginning to emerge to suggest that the dependence of some nekton species on the area of intertidal marsh or edge may not be as significant as once thought. Fry (2008) used shrimp density and stable isotope studies to show that open bays potentially support 50% of brown shrimp (BS; *Farfantepenaeus aztecus*) production in Barataria and Terrebonne Bays. If a large proportion of BS production can originate from open bays, Fry hypothesizes, the prediction of a fisheries collapse with increased marsh loss may not be realized.

In terms of the effects habitat alteration has on nekton populations in the western Gulf, much of the research has focused on penaeid shrimp and other invertebrates, such as blue crab (BC; *Callinectes sapidus*) at small spatial scales, which limits the applicability of the ecological dynamics at larger, estuary-wide scales (Browder et al. 1989, Minello et al. 1994, Zimmerman et al. 2002). While investigations on these species and at these spatial scales are valuable to understanding important ecological interactions, from a management perspective, it is important to assess the effect of marsh loss on the other species and on the food web as a whole. Fortunately, the Louisiana Department of Wildlife and Fisheries (LDWF) has kept a large fishery independent database since the late 1960's; a database such as this one is rare but extremely valuable to ecologists. Here I use these survey data to identify and describe patterns of long-term fish and shrimp abundance and habitat change (loss of marsh area and changes in the distance of marsh edge) for a number of commercially and recreationally important species in addition to investigating community-wide changes in Barataria Bay. The goal of this study is to test the following three null hypotheses:

H_0: Survey, landings and landings catch-per-unit-effort (CPUE) show similar patterns of abundance when regressed on marsh habitat (e.g. marsh area and edge);

H_0: Species abundance does not change after the maximum value of edge was achieved in 1985;

H_0: Community composition of estuarine nekton remained unchanged before, during and after the maximum value of edge was achieved in 1985.

To that end, I utilized commercial and recreational landings data in addition to 43 years of survey data from the LDWF monitoring program. Advances in GIS technology and the

increased availability of historical GIS data have allowed for more accurate estimates of the linear distance of marsh edge in Louisiana. The use of historical edge data and more accurate estimates of marsh area as calculated in GIS software will permit a more holistic analysis of past abundances that could potentially help managers determine if the null effect on Louisiana's fisheries is only temporary.

3.2. Methods

3.2.1. Study Area

The LA coast is separated into seven Coastal Study Areas (CSAs, Figure 3.1) to delineate the natural geographic changes that occur from east to west along the coast. This study focuses on CSA 3, which encompasses Barataria Bay, a 6280 km^2 sub-region of the Barataria-Terrebonne estuary system (Nelson et al. 2002). Barataria Bay (Figure 3.2), isolated from the MR (which is the easternmost boarder of the basin) since the 1940's, gets a majority of its freshwater input from rainfall and includes freshwater, brackish, and marine coastal marshes. Bayou LaFourche, an abandoned main channel of the MR, bounds the basin on west side (Conner and Day Jr 1987). In addition to rainfall, the Davis Pond Freshwater Diversion provides a controlled flow of freshwater into the upper reaches of the estuary.

3.2.2 Species of Interest

I used the same species here as in the analyses in Chapter 2; See Chapter 2 for a detailed description of the gear types used to capture the fish species used in this analysis. The two most abundant species in Barataria Bay are bay anchovy (BA; *Anchoa mitchilli*) and Atlantic croaker (AC; *Micropogonias undulates*) and will be referenced in the species biomass distribution (SBD) analyses.

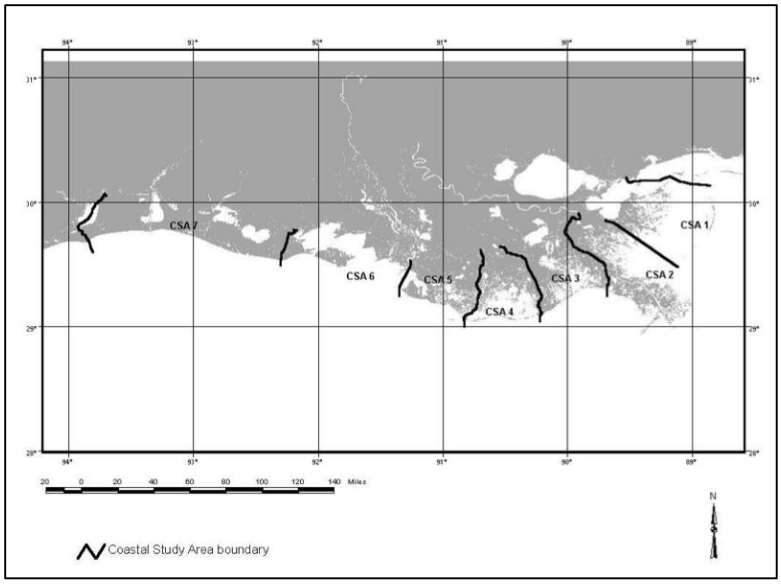

Figure 3.1. Louisiana's seven Coastal Study Areas. Barataria Bay is located solely within CSA 3.

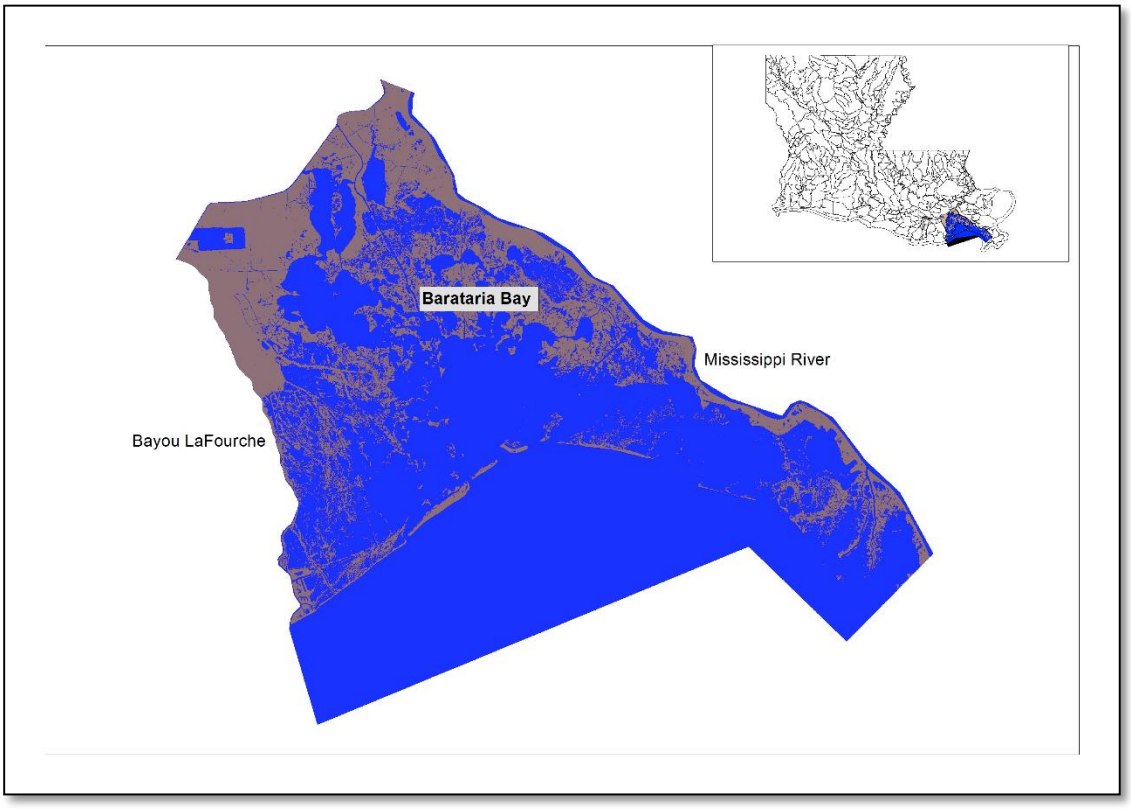

Figure 3.2. Barataria Bay, LA, USA, bordered on the east by the Mississippi River and on the west by Bayou LaFourche (Couvillion et al. 2011).

3.2.3 Data

The linear distance of marsh edge and the area of marsh were calculated using GIS data from Couvillion et al. (2011; Figure 3.3). For methods on how calculations for linear distance of marsh edge and marsh area were determined, please refer to Chapter 2.

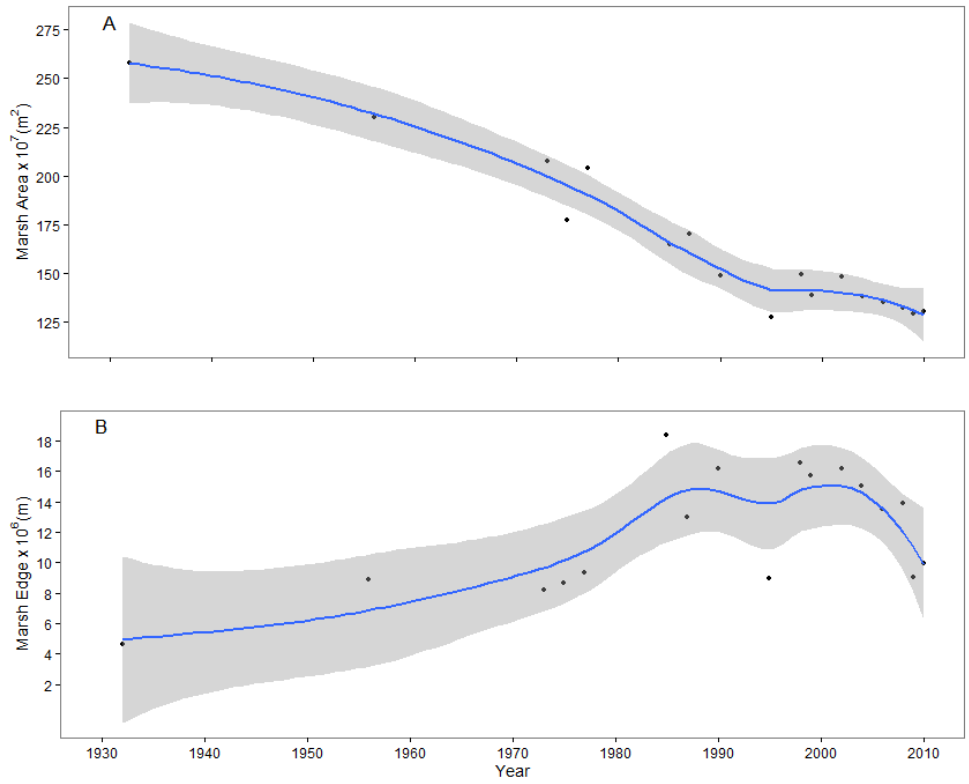

Figure 3.3. Habitat variables for Barataria Bay, LA from 1932-2010 (points) fitted with a Lowess curve (blue line with 95% confidence interval) to see the time trend. Marsh area (A) and marsh edge (B) in Barataria Bay were calculated with ArcGIS software from Couvillion (2011).

This study used both fishery independent (hereafter survey data) and fishery dependent catch (landings) data. For a detailed description of how these data were pre-processed for analysis, please see Chapter 2.

3.2.4. Statistical Analysis

I regressed survey data, landings data and landings CPUE data on area of marsh and edge to test the null hypothesis that survey and both types of landings data show similar patterns of abundance with habitat in Barataria Bay. For the regression plots, linear methods were used, but

if a linear relationship was not statistically significant, a smoothed, non-parametric curve was fit to the data using the Lowess method (Cleveland 1979). Exploration of the data using this non-parametric approach allowed for investigation of the "shape" of the curve between the habitats and species. For plots involving marsh area, the x-axes were inverted, with the highest values of marsh area closest to the origin. This approach to data visualization allowed for the relationship of marsh area to species abundance/landings to also represent what was truly occurring in nature (decreasing marsh over time).

To determine if trends in species survey data through time can be linked to the edge effect, as described by Browder et al. (1985, 1989), I first regressed the area of marsh on edge to determine the year in which the maximum value of marsh edge distance was observed. I next developed a series of regression trees for each species of interest, which is an extremely powerful method of non-parametric regression. This analysis recursively divides the predictor space into a number of regions, r, by minimizing the error sum of squares (Kutner et al. 2005). The regression trees were used to determine if there had been a change in the overall abundance patterns after the year in which the maximum value of edge was observed. There is potential to over fit regression trees; however the objective of this particular analysis was to identify any trends after the maximum value edge was observed, so over-fitting was not an issue of importance here. Regression tree analysis was conducted using the 'tree' package in the R statistical software (R 2013).

To further investigate Browder's conceptual model, I compared SBDs of Barataria Bay, before (1967-1969), during (1990-1992), and after (2007-2009) the maximum distance of edge was achieved. Using the statistical software Primer, three years of sample data for each time period were used in an Analysis of Similarity (ANOSIM), which is a non-parametric permutation

procedure used to test for differences among the three time periods (Clarke 1993). This procedure was performed on square-root transformed data using a Bray-Curtis similarity matrix. Significance for this analysis was set to an alpha level of 0.1%, similar to $\alpha = 0.01$ in a parametric statistical test. Next, to determine which species contributed most to the within group similarity and the between group dissimilarity, a Similarity Percentages analysis (SIMPER) was carried out. To perform the SIMPER analysis, the original sample data were first log (x + 1) transformed and a 90% cut-off percentage was specified, which stops listing species contributions once 90% of the similarity and dissimilarity is explained (Clarke 1993).

3.3. Results

3.3.1. Habitat variability analysis

Least squares regression for spotted seatrout (SStrout, *Cynoscion nebulosus)* catch indicated no significant linear relationship between SStrout survey or landings data with either marsh area or edge (Table 3.3). The fitted Lowess curves did not indicate any long-term trends, but there is an indication that SStrout abundance may increase over increasing values of marsh edge (Figure 3.4C). Spotted seatrout recreational landings remained relatively constant over most values of marsh area and edge (Figures 3.4B and D).

Table 3.3. Spotted seatrout CPUE ($lcpue_{sst}$) and SST recreational landings ($lrec\_lbs_{sst}$) regressed on area of marsh and edge. Response variables indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	**Predictor**	**Parameter Estimate**	F	p	$adj\ R^2$
$lcpue_{sst}$	$marsh$	-1.77e-07	0.02234	0.8825	-0.04247
$lrec\_lbs_{sst}$	$marsh$	-3.75e-07	2.511	0.1267	0.05923
$lcpue_{sst}$	$edge$	2.52e-05	0.1249	0.727	-0.03784
$lrec\_lbs_{sst}$	$edge$	-2.39e-05	2.854	0.1046	0.07172

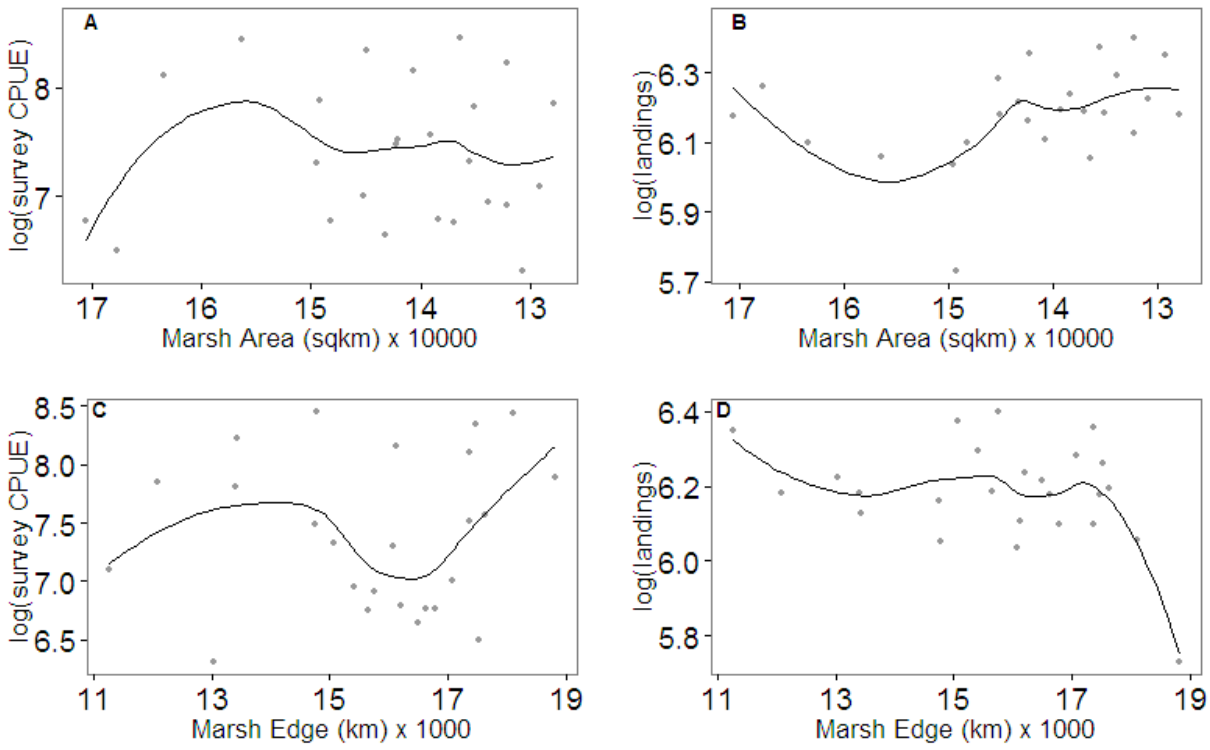

Figure 3.4. Habitat change plots for spotted seatrout (*Cynoscion nebulosus*) with Lowess fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of recreational landings and marsh area; (C) log of survey CPUE and edge; (D) log of recreational landings and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss).

Least squares regression of red drum (RD; *Sciaenops ocellatus)* survey and landings data on marsh area and edge indicates a linear relationship exists for all but one relationship tested (Figure 3.5). Red drum relative abundance and landings regressions both indicated that RD abundance increases as marsh area decreases (Figure 3.5A-B). Results of the regressions with edge showed a no effect on relative abundance of RD, while landings data showed a drastic decrease as edge increases (Table 3.4, Figure 2.5C-D).

Table 3.4. Red drum (*Sciaenops ocellatus*) CPUE ($lcpue_{RD}$) and red drum recreational landings ($lrec\_lbs_{RD}$) regressed on area of marsh and edge. Response variables indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	Predictor	Parameter Estimate	F	p	R^2
$lcpue_{RD}$*	$marsh$	-3.716e-06	6.02	0.02214	0.173
$lrec\_lbs_{RD}$*	$marsh$	-1.366e-06	31.79	<.0001	0.5619
$lcpue_{RD}$	$edge$	-1.297e-04	1.717	0.203	0.02901
$lrec\_lbs_{RD}$*	$edge$	-5.563e-05	8.291	0.0085	0.233

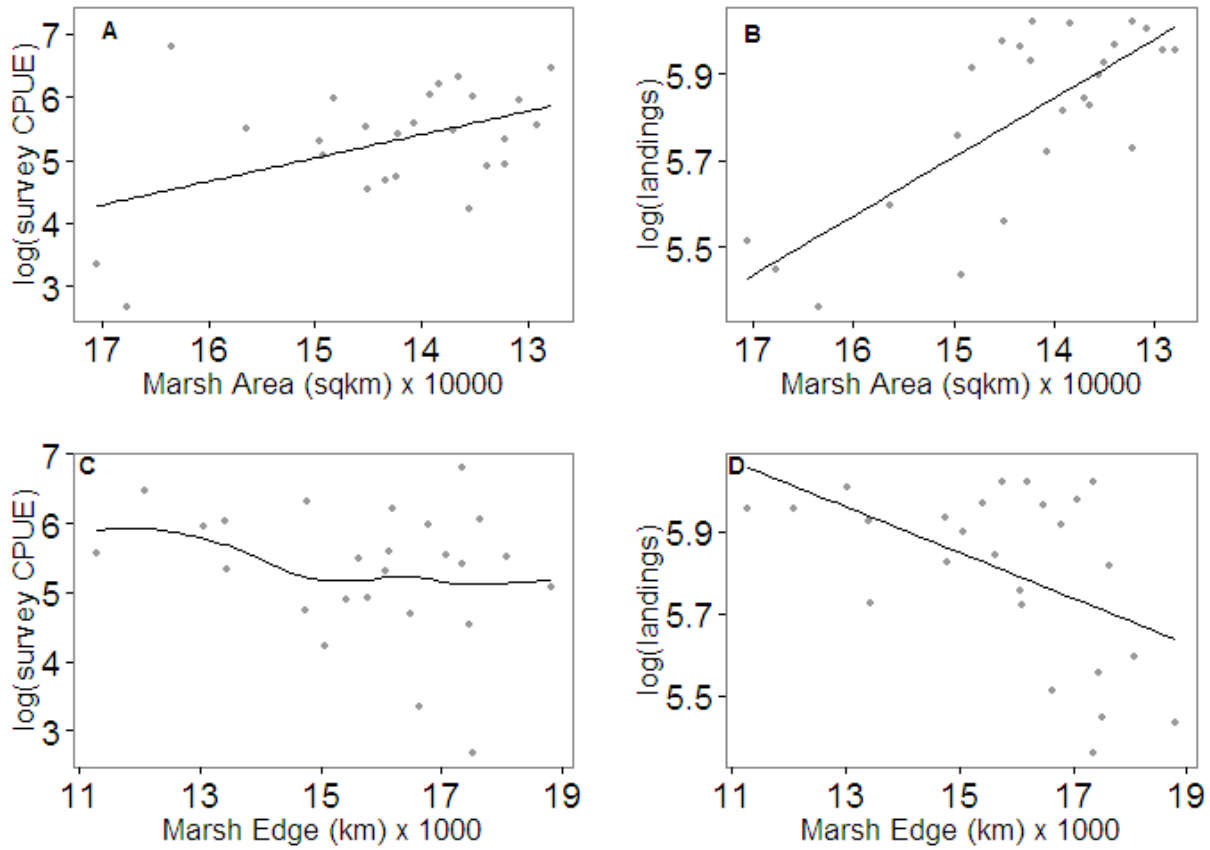

Figure 3.5. Habitat change plots for red drum (Sciaenops ocellatus) with fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of recreational landings and marsh area; (C) log of survey CPUE and edge; (D) log of recreational landings and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss)

Least squares regression of Gulf menhaden (GuM, *Brevoortia patronus)* on marsh area and edge suggest contrasting responses between survey and landings data (Table 3.5). As marsh area decreases, relative abundance of GuM showed a decreasing trend, while menhaden landings data showed a parabolic relationship, with a maximum catch of approximately 1.75 x 10^5 km^2 (Figure 3.6A-B). Regression of GuM survey data on edge showed a significant negative trend as edge increases (Table 3.4), while analysis of landings data on edge presented relatively consistent variation in catch over most values of edge (Figure 3.6C-D).

Table 3.5. Gulf menhaden (*Brevoortia patronus*) CPUE ($lcpue_{GM}$) and Gulf menhaden commercial landings ($lcomm\_lbs_{GM}$) regressed on area of marsh and edge. Response variables indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	**Predictor**	**Parameter Estimate**	F	p	R^2
$lcpue_{GM}$*	$marsh$	3.057e-06	55.92	<.0001	0.5609
$lcomm\_lbs_{GM}$	$marsh$	1.262e-08	0.03822	0.8459	-0.0229
$lcpue_{GM}$*	$edge$	-2.073e-04	27.33	<.0001	0.3798
$lcomm\_lbs_{GM}$	$edge$	8.729e-06	2.935	0.0941	0.0431

Least squares regression of BS survey and landings data with marsh area and edge indicated no significant linear relationships, but when landings CPUE data was regressed on either habitat type, contrasting, significant trends were apparent (Table 3.6). When BS abundance data were regressed on marsh area, three distinct and contrasting patterns of abundance occurred over decreasing values of marsh area (Figure 3.7A-C). BS data were regressed on edge, survey and landings data indicated a positive parabolic trend, while landings CPUE showed a sharp decline in abundance with increasing values of edge (Figure 3.7D-F).

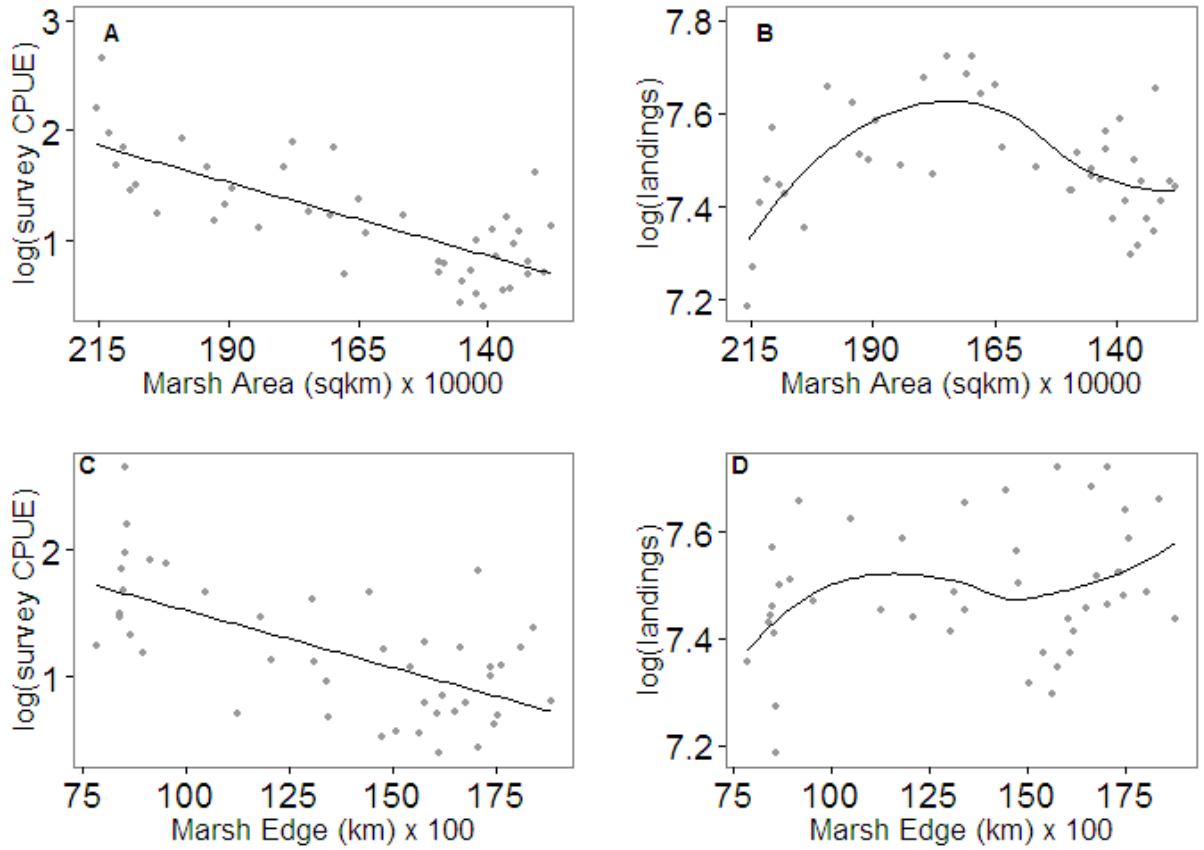

Figure 3.6. Habitat change plots for Gulf menhaden (*Brevoortia patronus*) with fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of commercial landings and marsh area; (C) log of survey CPUE and edge; (D) log of commercial landings and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss)

Table 3.6. Brown shrimp survey CPUE ($lcpue_{BS}$), brown shrimp (*Farfantepenaeus aztecus*) commercial landings ($lcomm\_lbs_{BS}$), and brown shrimp landings CPUE ($lcomm\_cpue_{BS}$) regressed on area of marsh and edge. Response variables indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	Predictor	Parameter Estimate	F	p	R^2
$lcpue_{BS}$	$marsh$	5.301e-07	2.927	0.0945	.043
$lcomm\_lbs_{BS}$	$marsh$	6.427e-08	1.296	0.2613	0.007
$lcomm\_cpue_{BS}$*	$marsh$	-3.706e-06	14.79	.004	.58
$lcpue_{BS}$	$edge$	-2.969e-05	1.33	0.2554	0.008
$lcomm\_lbs_{BS}$	$edge$	6.525e-06	2.04	0.1606	0.024
$lcomm\_cpue_{BS}$*	$edge$	-1.345e-04	20.99	.001	.67

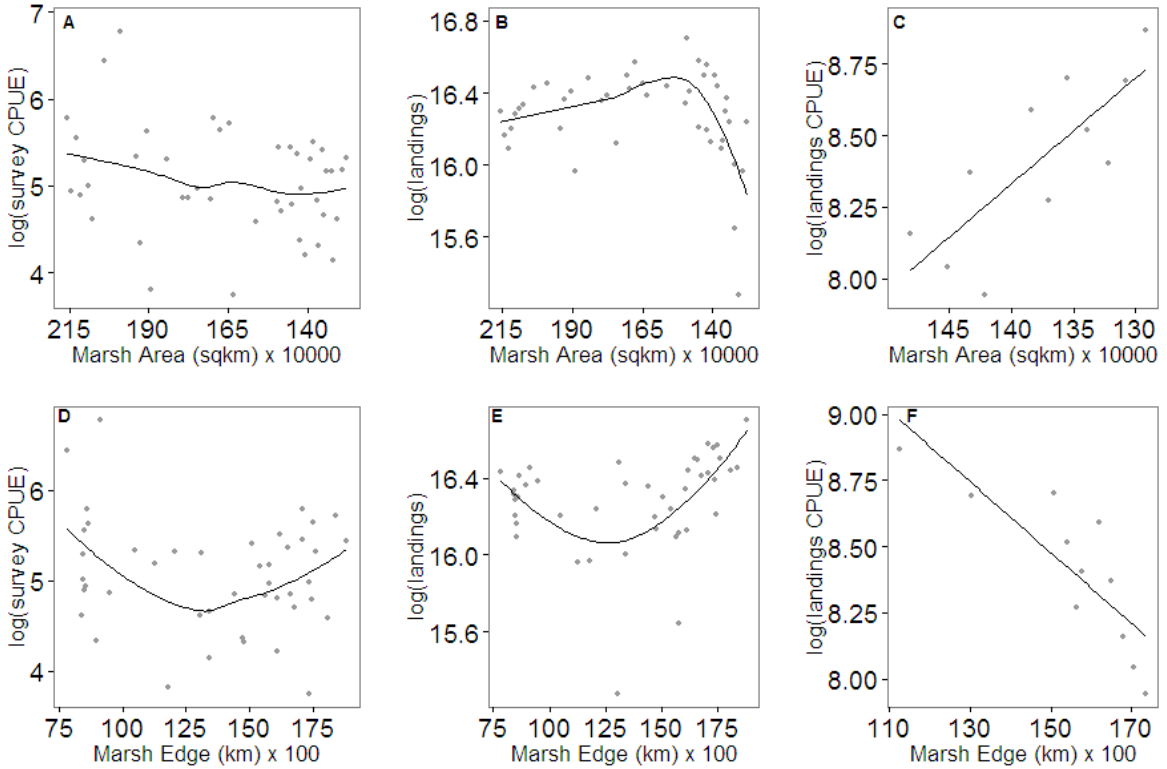

Figure 3.7. Habitat change plots for brown shrimp (*Farfantepenaeus aztecus)* with fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of commercial landings and marsh area; (C) log of landings CPUE and marsh edge distance; (D) log of survey CPUE and edge; (E) log of commercial landings and edge; (F) log of landings CPUE and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss

Least squares regression of white shrimp (WS, *Litopenaeus setiferus)* on marsh area and edge suggested varied responses (Table 3.7). While survey data of WS showed no significant relationship with marsh area, WS landings and landings CPUE show a significant increasing trend as marsh area decreases (Figure 3.8A-C). White shrimp survey data appear constant over most values of edge; however, landings data indicated a significant increasing trend, while landings CPUE showed a highly significant decreasing trend (Figure 3.8D-F).

Least squares regression of BC on marsh area and edge suggested varied responses. The results indicated significant relationships between BC abundance and marsh area (Table 3.8, Figure 3.9A-C). Regressions of BC abundance on edge showed varied results (Figure 3.9D-E). Relative abundance of BC did not exhibit a linear trend over increasing values of edge, but a slight positive parabolic curve can be observed. Blue crab landings data, when regressed on edge, indicated a significant increasing trend, while landings CPUE indicated a significant decreasing trend.

Table 3.7. White shrimp *(Litopenaeus setiferus)* survey CPUE ($lcpue_{WS}$), white shrimp commercial landings ($lcomm\_lbs_{WS}$), and white shrimp landings CPUE ($lcomm\_cpue_{WS}$) regressed on area of marsh and edge Response variables indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	**Predictor**	**Parameter Estimate**	F	p	R^2
$lcpue_{WS}$	$marsh$	1.251e-07	0.060	0.806	-0.022
$lcomm\_lbs_{WS}$*	$marsh$	-2.218e-07	16.27	0.0002	0.262
$lcomm\_cpue_{WS}$*	$marsh$	-4.415e-06	18.93	0.002	0.64
$lcpue_{WS}$	$edge$	-4.552e-05	1.246	0.271	0.006
$lcomm\_lbs_{WS}$*	$edge$	1.353e-05	7.758	0.008	0.136
$lcomm\_cpue_{WS}$*	$edge$	-0.0001237	7.49	0.02	0.39

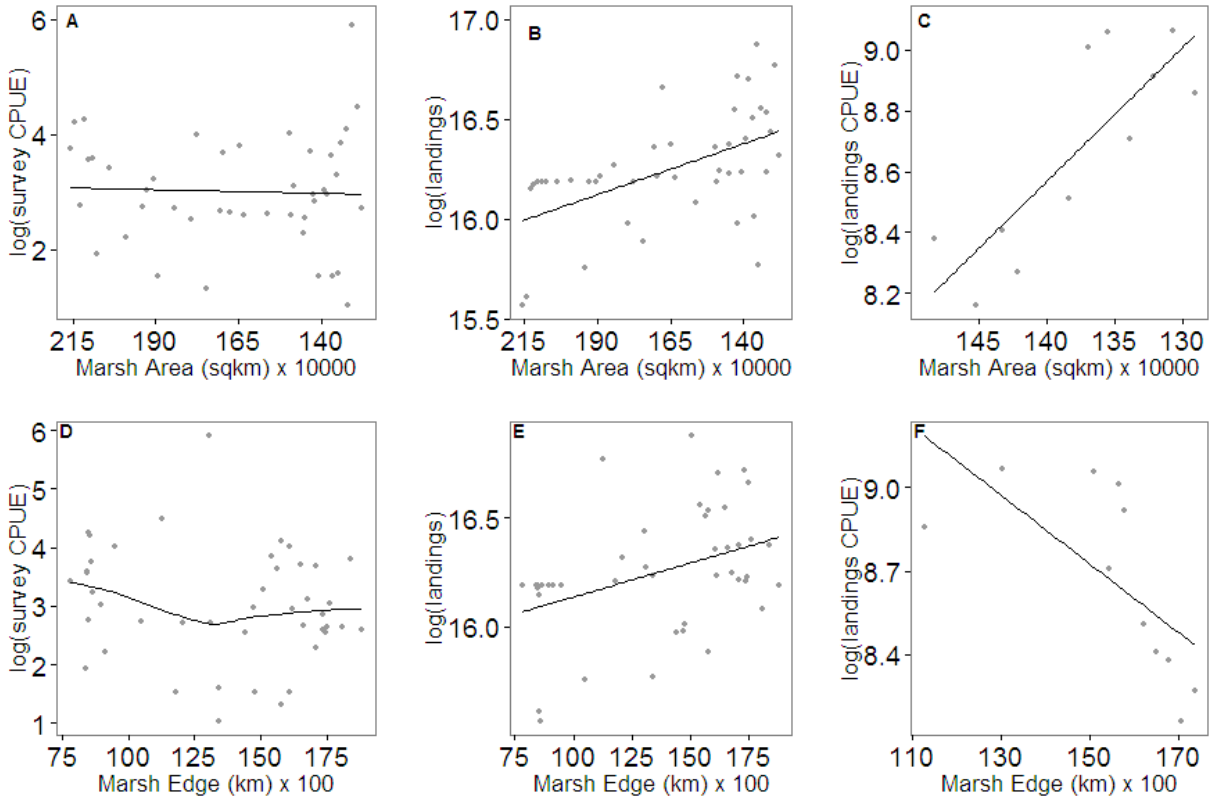

Figure 3.8. Habitat change plots for white shrimp *(Litopenaeus setiferus)* with fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of commercial landings and marsh area; (C) log of landings CPUE and edge; (D) log of survey CPUE and edge; (E) log of commercial landings and edge; (F) log of landings CPUE and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss)

Table 3.8. Blue crab (*Callinectes sapidus*) survey CPUE ($lcpue_{BC}$), blue crab commercial landings ($lcomm\_lbs_{BC}$), and blue crab landings CPUE ($lcomm\_cpue_{BC}$) regressed on area of marsh and edge. Species indicated with an (*) a significant time trend in the data ($\alpha < .05$).

Response	Predictor	Parameter Estimate	F	p	R^2
$lcpue_{BC}$*	$marsh$	1.397e-06	14.36	0.0005	0.237
$lcomm\_lbs_{BC}$*	$marsh$	-7.173e-07	83.38	<.0001	0.657
$lcomm\_cpue_{BC}$*	$marsh$	-0.0001237	7.49	0.02	0.39
$lcpue_{BC}$	$edge$	-4.664e-05	1.87	0.1788	0.0198
$lcomm\_lbs_{BC}$*	$edge$	5.599e-05	65.18	<.0001	0.5988
$lcomm\_cpue_{BC}$*	$edge$	-5.959e-05	6.02	0.04	0.33

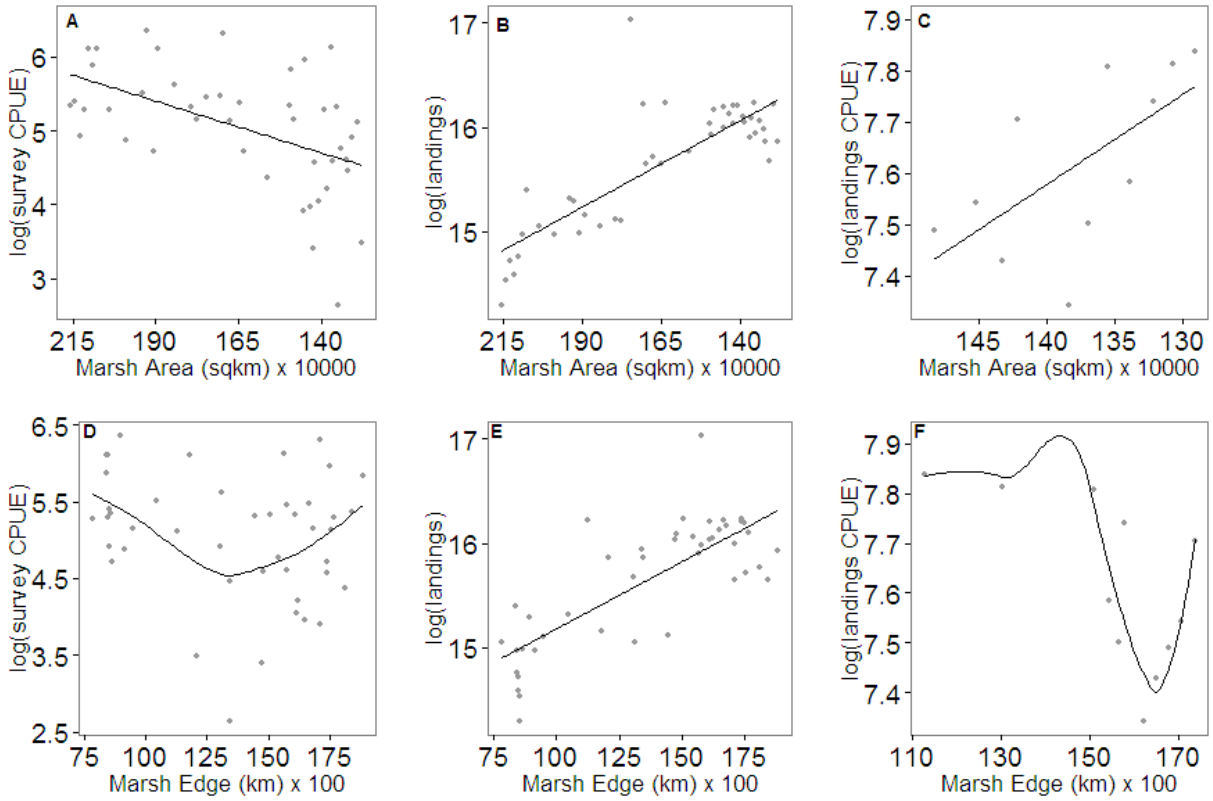

Figure 3.9. Habitat change plots for blue crab (*Callinectes sapidus)* with fitted regression lines. (A) log of survey CPUE and marsh area; (B) log of commercial landings and marsh area; (C) log of landings CPUE and edge; (D) log of survey CPUE and edge; (E) log of commercial landings and edge; (F) log of landings CPUE and edge. (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss)

3.3.3 Edge Effect

This study found that the maximum value of edge was achieved in 1985, 10 years prior to estimates by Browder et al. (1989, Figure 3.10). The findings of this study empirically corroborates the theory that marsh area, when plotted on edge, follows a dome-shaped curve (Browder et al. 1985, Browder et al. 1989, Chesney et al. 2000, Zimmerman et al. 2002). Using the year 1985 as a significant event, tree regressions indicated that for BS WS and BC, decreases in mean abundance began occurring after 1985 with lower values continuing for 5-7 years. After that time, abundance levels began a gradual recovery to pre-1985 levels (Figures 3.11-3.13). Gulf menhaden survey data show a generally stable abundance for the 15 years prior to 1985 but

a steady decrease in GuM relative abundance can be seen beginning around 1985. After the observed decrease, GuM relative abundance becomes relatively stable but at levels much less than pre-1985 (Figure 3.14). Spotted seatrout survey data increased quite rapidly after 1985, but beginning around 1995 overall abundance of SStrout declined (Figure 3.15). Red drum abundance shows an increasing trend post-1985 and then abundance becomes generally stable over time (Figure 3.16).

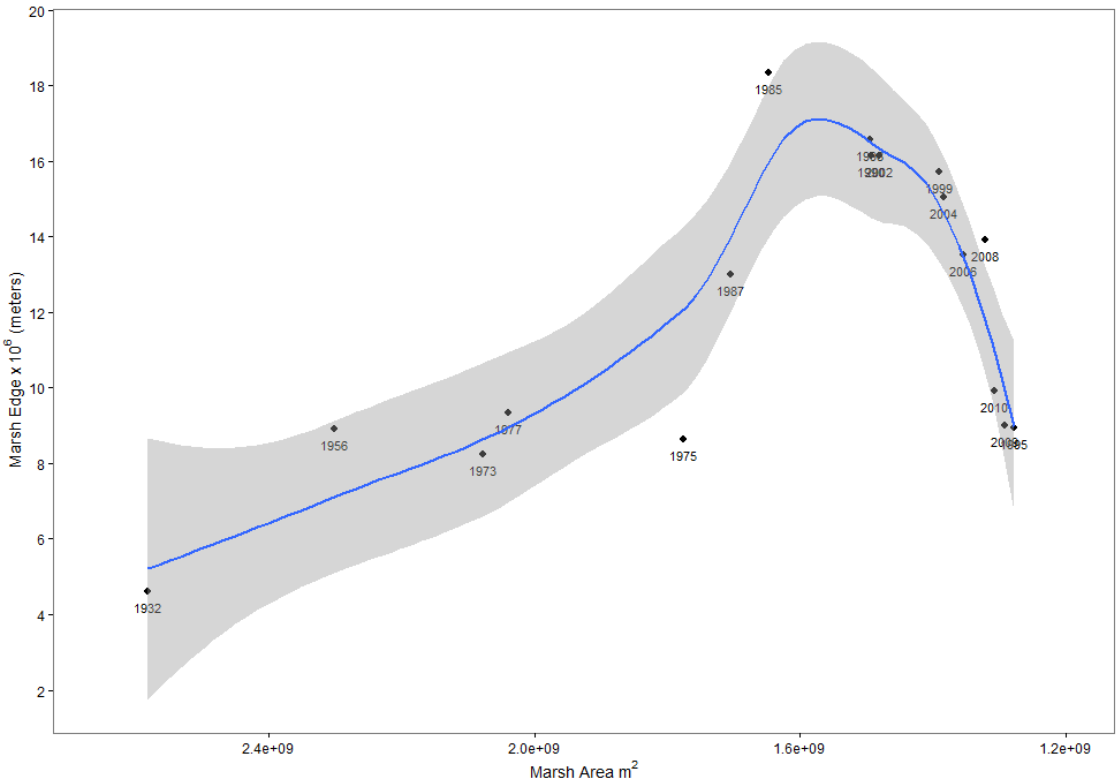

Figure 3.10. Lowess regression of marsh area and edge from Barataria Bay from 1932 to 2010 (blue line with 95% confidence interval). The maximum linear distance of marsh edge was achieved at ~1.9 x 10^7 m in 1985, all values of edge after 1985 decreased from that apex. Points indicate values of edge and marsh area derived from Couvillion et al (2011). (Note the reverse x-axis on the marsh area plots, representing the temporal component of marsh loss)

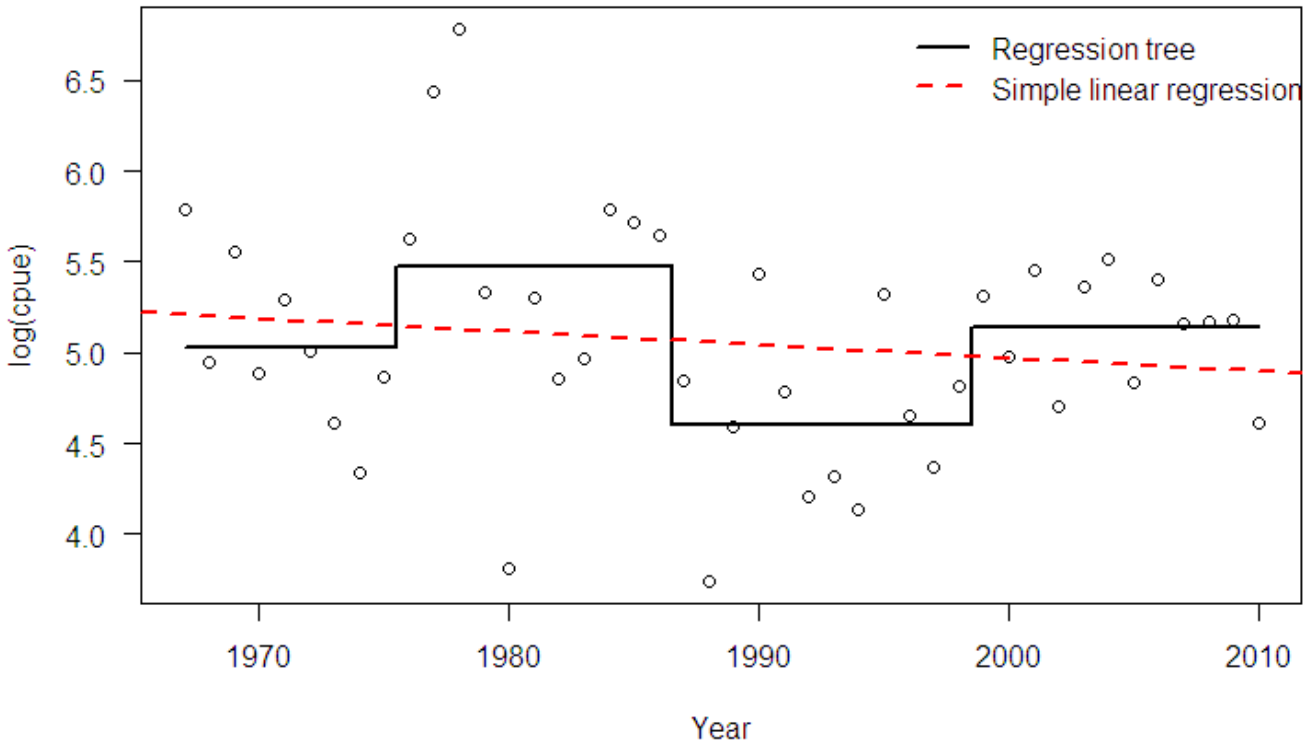

Figure 3.11. Simple linear regression (SLR) and tree regression of brown shrimp (*Farfantepenaeus aztecus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

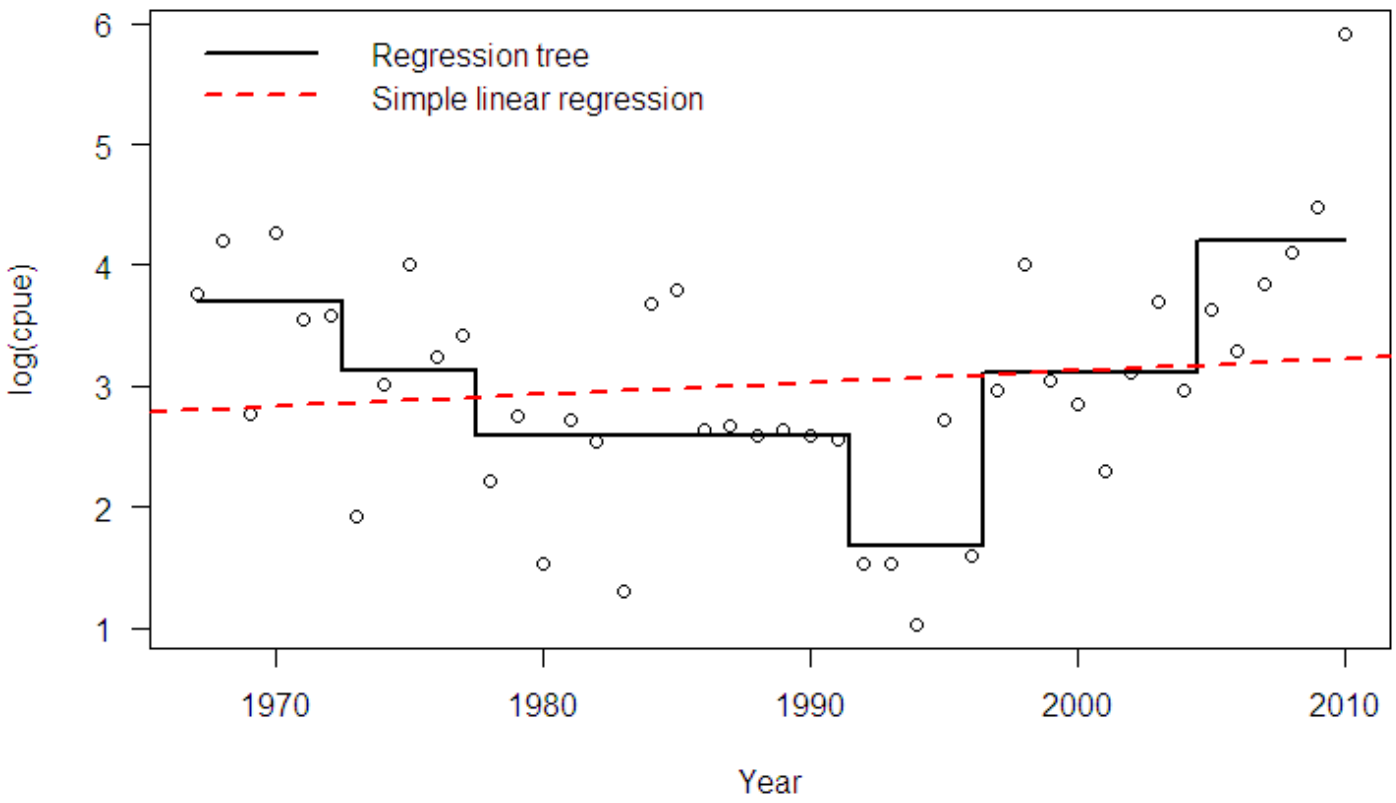

Figure 3.12. Simple linear regression and tree regression of white shrimp (*Litopenaeus setiferus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

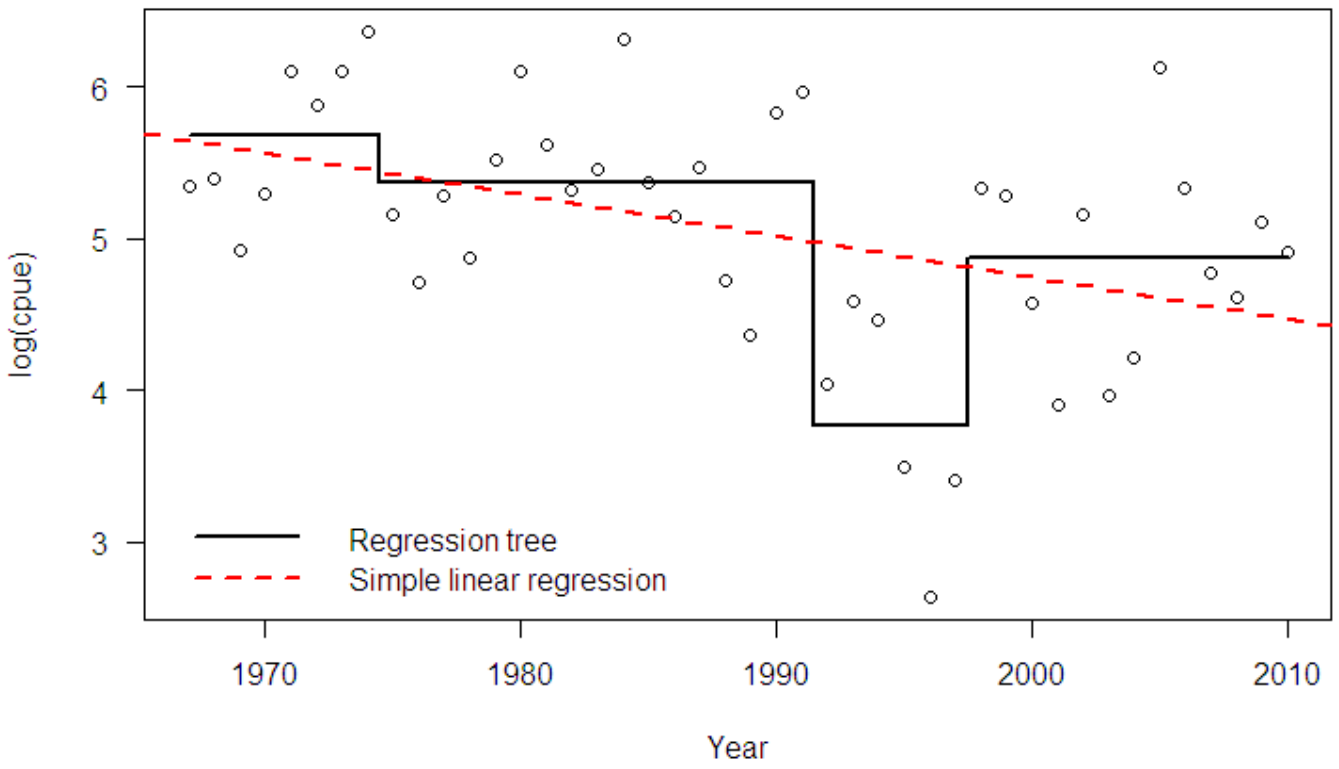

Figure 3.13. Simple linear regression and tree regression of blue crab (*Callinectes sapidus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

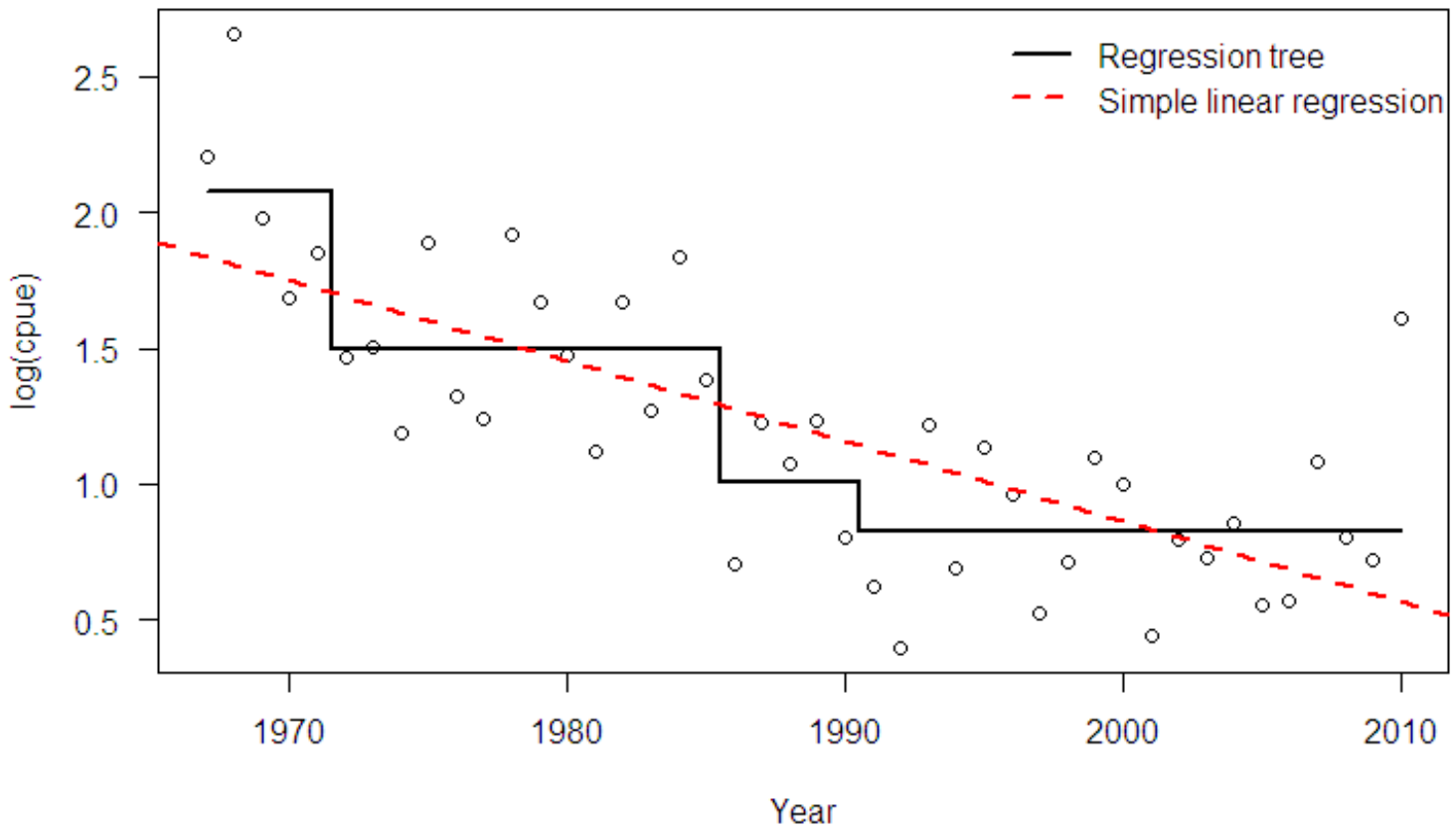

Figure 3.14. Simple linear regression and tree regression of Gulf menhaden (*Brevoortia patronus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

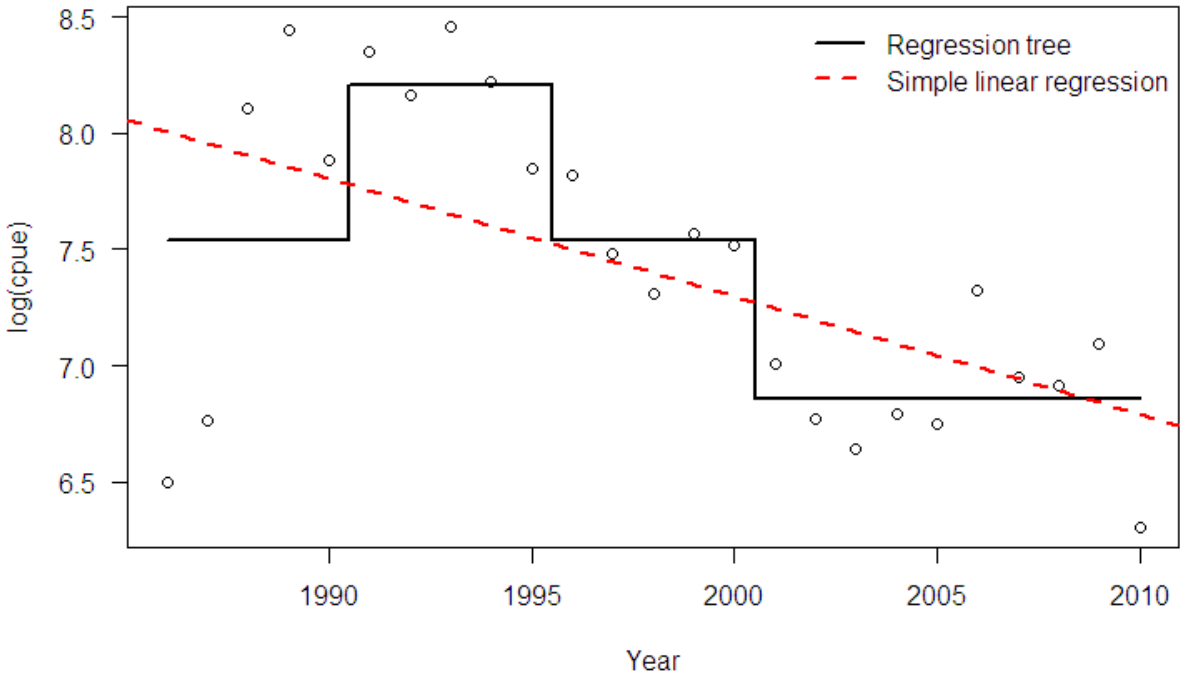

Figure 3.15. Simple linear regression and tree regression of spotted seatrout (*Cynoscion nebulosus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

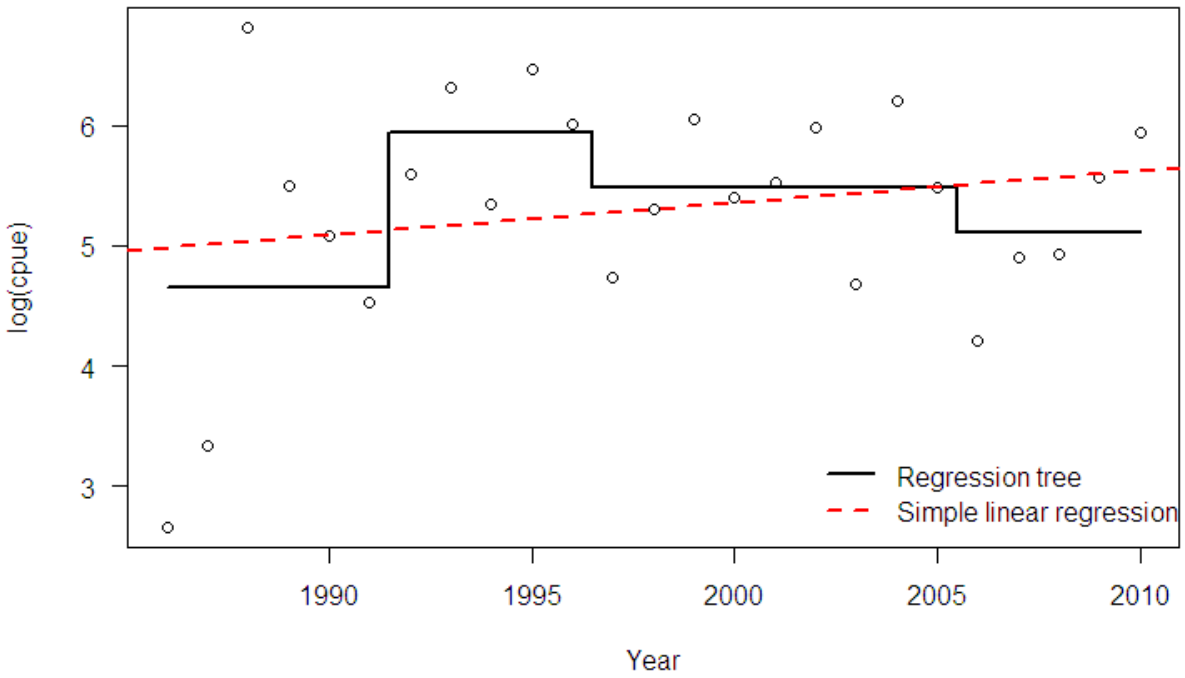

Figure 3.16. Simple linear regression and tree regression of red drum (*Sciaenops ocellatus*) survey data with time. The solid black line represents the results of the tree regression, while the dashed red line represents the SLR.

Results of the ANOSIM found a highly significant difference between all time periods (R = 0.061, p = 0.001, Table 3.8). Although the R-statistic is relatively low, when sample numbers are as high as there are in this study (N = 1889), significance can still be found (Clarke 1993). All pairwise comparisons are statistically different from one another, but the low R-values found in theses pairwise comparisons indicate there is high overlap between many of the species in the groups being compared (McCawley 2003).

Table 3.8. Global and pairwise R-statistic values comparing the community structure of Barataria Bay at three different time periods.

Global ANOSIM: R = 0.061, p = .001

	Time 1	Time 2	Time 3
Time 1	-	-	-
Time 2	0.05	-	-
Time 3	0.085	0.05	-

Results of the SIMPER analysis show that species that contribute most to within in time period similarity remains relatively constant over all three groups. In time periods 1 and 2, BA were the only species consistently caught in all samples, which is determined by values of Sim/SD > 1 (Table 3.9). The percent contribution of each species in caught in each sample, however, changes considerably from one time period to the next. While BA and AC are clearly the most abundant species in Barataria Bay, their presence in samples seems to generally decline from Time 1 to Time 3. The community in Time 1 is dominated by BA and AC, however, Time 3 shows that the nekton community is becoming more diverse and less dominated by a just a few taxa. The largest contribution of dissimilarity of pairwise time period comparisons were from BA and AC due to their high abundance in the estuary (Table 3.10). For all three time periods, BA and AC are most responsible for distinguishing between the two time periods being compared (Diss/SD > 1).

Table 3.9. Results of the SIMPER analysis showing the species that contribute most to the within time period similarity. Total average similarity between samples for each time period is shown in bold. Sim/SD is the ratio of average similarity and the standard deviation—values above 1 contribute largely the similarity within groups. % Contri. is the percent similarity each speices is responsible for within each time period, while % Cum. Contri. is the cumulative percent contribution to similarity. Bay anchovy (*Anchoa mitchilli*) Atlantic croaker (*Micropogonias undulates*), brown shrimp (*Farfantepenaeus aztecus),* white shrimp *(Litopenaeus setiferus),* blue crab *(Callinectes sapidus),* Gulf menhaden (*Brevoortia patronus).* spot (*Leiostomus zanthurus*), sea catfish (*Arius felis*), Atlantic brief squid (*Lolliguncula brevis),* least puffer (*Sphoeroides parvus*), fringed flounder (*Etropus crosstus*), sand seatrout (*Cynoscion arenarius*), bighead sea robin (*Pronotus tribulus*).

Time Period	Species	Avg. Abun.	Avg. Sim.	Sim/SD	% Contri.	% Cum. Contri.
Time 1			**33.71**			
	Bay anchovy	3.95	14.98	1.09	44.45	44.45
	Atlantic croaker	2.11	5.45	0.84	16.16	60.61
	Blue crab	1.12	3.11	0.78	9.24	69.84
	Brown shrimp	1.38	2.36	0.49	7.01	76.85
	Gulf menhaden	1.26	2.21	0.49	6.54	83.40
	Spot	0.89	1.22	0.44	3.61	87.01
	Sea catfish	0.69	0.99	0.38	2.94	89.95
	White shrimp	0.54	0.69	0.27	2.05	92.00
Time 2			**25.83**			
	Bay anchovy	3.20	13.08	1.02	50.62	50.62
	Brown shrimp	1.28	2.84	0.50	10.99	61.61
	Atlantic croaker	1.19	2.17	0.46	8.41	70.02
	Blue crab	0.86	1.60	0.46	6.19	76.21
	Atlantic brief squid	0.67	1.24	0.33	4.78	81.00
	White shrimp	0.65	1.08	0.38	4.20	85.19
	Sea catfish	0.48	0.60	0.25	2.33	87.53
	Least puffer	0.31	0.37	0.24	1.14	88.94
	Fringed flounder	0.26	0.31	0.22	1.21	90.15
Time 3			**27.49**			
	Bay anchovy	2.72	8.63	0.91	31.38	31.38
	Atlantic croaker	2.02	5.22	0.76	18.98	50.36
	Brown shrimp	1.50	3.68	0.66	13.39	63.75
	White shrimp	1.34	3.29	0.68	11.96	75.71
	Blue crab	0.77	1.66	0.51	6.05	81.76
	Atlantic brief squid	0.42	0.58	0.27	2.12	83.88
	Sand seatrout	0.43	0.49	0.29	1.80	85.68
	Least puffer	0.38	0.47	0.29	1.70	87.38
	Bighead sea robin	0.41	0.39	0.28	1.44	88.82
	Spot	0.41	0.37	0.24	1.33	90.15

Table 3.10. Results of the SIMPER analysis showing the species that contribute most to the dissimilarity between all pairwise comparisons. Total average dissimilarity between time periods are shown in bold. Diss/SD is the ratio of average dissimilarity and the standard deviation—values above 1 contribute largely the dissimilarity within groups. % Contri. is the percent dissimilarity each speices is responsible for in each time period, while % Cum. Contri. is the cumulative percent contribution to similarity. Bay anchovy (*Anchoa mitchilli*) Atlantic croaker (*Micropogonias undulates*), brown shrimp (*Farfantepenaeus aztecus),* white shrimp *(Litopenaeus setiferus),* blue crab *(Callinectes sapidus),* and Gulf menhaden (*Brevoortia patronus).*

Time Period	Species	Avg. Abun.	Avg. Abu	Avg. Diss	Diss/SD	% Contr.	% Cum. Contri.
2 vs 3		Time 2	Time 3	**77.51**			
	Bay anchovy	7.94	6.67	17.08	1.14	22.03	22.03
	Atlantic croaker	2.11	4.3	8.49	0.97	10.95	32.98
	Brown shrimp	2.27	2.76	7.09	0.82	9.15	42.13
	White shrimp	0.94	2.23	4.64	0.8	5.98	48.11
	Blue crab	1.28	1.04	3.39	0.79	4.38	52.49
2 vs 1		Time 2	Time 1	**75.76**			
	Bay anchovy	7.94	12.31	19.84	1.22	26.19	26.19
	Atlantic croaker	2.11	4.38	7.61	0.99	10.05	36.24
	Brown shrimp	2.27	2.75	6.37	0.8	8.4	44.64
	Gulf menhaden	0.11	2.43	4.11	0.53	5.43	50.07
	Blue crab	1.28	1.56	3.47	0.89	4.59	54.66
3 vs 1		Time 3	Time 1	**76.55**			
	Bay anchovy	6.67	12.31	19.25	1.19	25.14	25.14
	Atlantic croaker	4.3	4.38	8.85	1.05	11.56	36.7
	Brown shrimp	2.76	2.75	6.46	0.86	8.44	45.14
	White shrimp	2.23	0.83	4.15	0.7	5.42	50.56
	Gulf menhaden	0.16	2.43	4.02	0.53	5.25	55.82

3.4 Discussion

The Browder et al. (1989) study found a significant positive linear relationship between increasing distance of land-water interface (edge) and BS catch, but their results are generally contradictory to the findings in this study. Comparisons of BS and WS landings data with the edge showed a slight positive relationship exists. However, when effort is considered with landings data, a strong, significant negative relationship is observed for both BS and WS. These two contradicting analyses further boasts support that using landings data as part of habitat comparisons is Louisiana can be problematic, if not completely incorrect because survey-marsh edge regressions show a null effect on penaeid shrimp abundance over most values of edge. Another possible explanation as to the differences in trends found in this study as compared to Browder et al. (1989), is the simple fact that their marsh-water interface measurements were simulated values rather than the more precise values estimated with the Couvillion et al. (2011) dataset. I contend, however, that the differences seen in this study when compared with previous studies are more complicated. For instance, it is unlikely that all members of a population are found at or near the edge, which suggests that nekton production is influenced by other factors or mechanisms, regardless of how much edge is available for organisms (Fry 2008). With the combination of marsh loss and lack of fluvial input into Barataria Bay, long-shore currents flowing from east to west could also be responsible for transporting nutrients and phytoplankton from the Mississippi River into the estuary, benefiting estuarine organisms and counter-acting negative impacts of marsh loss. More likely is that the resilience seen in many of these estuarine species could be derived from their adaptations to dynamic ecosystems over thousands of years. Even though the delta cycle has effectively stopped in coastal LA, it is clear that habitat change is still occurring, albeit at potentially higher rates because of the synergistic effects of multiple

perturbations. Estuarine species may already be well adapted for a constantly changing ecosystem, and in this case, land loss may not affect nekton as severely as once hypothesized. While determining exactly what is driving secondary production in Barataria Bay is outside the scope of this study, the results herein can help guide future studies to determine potential mechanisms that drive long-term variability in nekton abundance.

If Browder's conceptual model is indeed correct, the loss of edge in the marsh since 1985 should have resulted in decreased shrimp biomass in Louisiana. Browder et al. (1989) hypothesized that once edge length begins to decline after achieving a maximum (when marsh and water area are both 50% of the total area of the basin), BS yields will also begin to decline. The estimated time of the decline in marsh-water interface in Browder et al. (1989) was 1995. This study found that the maximum edge distance occurred in 1985, earlier than was expected by their study (Figure 3.10). Tree regressions suggest that for some species (BS, WS, BC), a short decline in abundance was observed after 1985. More interesting were these species' ability to recover rather quickly (~5-7 years) after the historical maximum value of marsh edge distance was achieved in 1985. The mechanisms by which these species have adjusted to habitat loss in the bay are unclear, but one study suggests that increases in fisheries production, even considering the loss of habitat, could be a result of estuarine-like conditions on the shallow shelf during periods of increased flow from the Mississippi River (Cowan et al. 2008). This hypothesis is corroborated by the fact that all three of these species move off-shore at some point in their life history. Coupled with stable isotope studies conducted by Fry (2008) in Barataria and Terrebonne Bays, it seems likely that penaeid shrimp production benefits from many other factors from both inside and outside of the estuary proper. And, as previously stated, these species have existed in a deltaic system for thousands of years, and may have evolved the

capability to deal with dynamic changes in habitat, whether caused by delta switching or leveeing of the MR. It has been almost 30 years since the maximum value of edge occurred in Barataria Bay and the results of this study show that the predicted decline in estuarine-dependent species after the marsh becomes more than 50% water, may have been averted, at least for penaeid shrimp and BC. As with every study involving organisms that are exploited in fisheries, the results herein could also be (and most likely are) confounded by fishing pressure within the ecosystem. Over the same period of time since marsh edge reached its maximum, substantial decreases in shrimping effort have occurred. Factors such as the import of Asian shrimp, rises in fuel prices, and loss of infrastructure from hurricanes have facilitated this decrease in effort. Because of the effects of fishing with in the system, disentangling the mechanisms driving population variations over time becomes difficult. Further studies will be needed that take into account fishing pressure and these efforts are already under way by this author.

Nekton species have been historically positively correlated with marsh area (Turner 1977, Zimmerman et al. 2002, Haas et al. 2004, Valiela et al. 2004), but this study challenges some of those trends. Turner (1977) found a strong positive linear relationship with penaeid shrimp landings and increasing area of marsh, but when the penaeid species were separated for this analysis, I found WS landings to increase with *decreasing* marsh area. On the other hand, BS landings indicated a no effect on catch up to a point at which landings decrease rapidly with decreasing marsh area. When effort is included in the landings data/marsh area regressions (Figures 3.7C and 3.8C), the results show that penaeid shrimp abundance increases with decreasing values of marsh area. The differing trends in this analysis when compared to previous studies could be explained by a number of factors, one being that the data were too aggregated by both region and species to represent actual trends in this variable system. At this

level of aggregation, patterns in data can be lost or results might suggest relationships that are the opposite of what is truly occurring in nature (Simpson's Paradox). Further, using landings data as a proxy for abundance requires an assumption of constant fishing mortality and catchability in the catch equation:

$$C = qfN; \qquad \text{Eq. 3.1}$$

where C is catch or landings, N is the number of species in the population, f is the fishing effort and q is the catchability coefficient. With the knowledge we have in LA about changes in fishing effort and changes in fishing practices, assuming constant fishing mortality and catchability is too far-reaching. As previously mentioned, shrimping effort has decreased over the last 20 years. If abundance has remained stable as indicated by analyses on survey data and marsh area, then it could be hypothesized that the fishermen left in the industry are actually being more effective at catching their target species—thus increasing the overall catch during each trip. Moreover, habitat changes in Barataria Bay are occurring rapidly and these changes are more than likely affecting fishing patterns for commercial vessels, allowing vessels to access parts of the marsh that they historically have not be able to navigate. It must be emphasized that neither comparisons of marsh area with landings CPUE data or landings data support the findings of the marsh area-survey data analyses for brown or white shrimp. Even though landings of BS begin to decrease over lower values of marsh area, the decrease in landings could be confounded by economic factors such as the decrease of shrimp prices to due competition with Asian markets. For WS, if landings data were used to summarize the relationship of this species with area of marsh, one could conclude, perhaps incorrectly, that WS abundance increases significantly with less marsh area. It is more believable, as the survey data indicate, that both BS and WS abundance have been generally robust to changes in marsh area.

Previous studies indicated that juvenile GuM were associated with edge habitat, while other studies did not show such an association (Baltz et al. 1993, Rozas and Reed 1993, Rozas et al. 2007). The results of this study could likely support either of the previous findings mentioned. I found that as edge increases, survey data of GuM decreases. With the known association of juvenile GuM with marsh edge as per Baltz et al. (1993), it could be hypothesized that as edge increases GuM relative abundance would also increase. This hypothesis is unsupported by the findings in this study (Figure 3.6A). Disparities between these findings and Baltz et al. (1993) could arise from a number of issues such as gear type used to collect samples, duration of the study (2 years vs. 44 years), or the location of the study area (samples were collected in a limited geographic area rather than being collected over all of Barataria Bay). Baltz et al. (1993) provided an interesting caveat to their findings. They hypothesized that edge in particular may not be as important to nekton species but rather the edge acts indirectly as a pathway by which organisms can access the internal area of the marsh when it is inundated. If this hypothesis is true, then comparisons between GuM relative abundance and marsh area begins to gain some traction because survey data decrease with decreasing values of marsh area (Figure 3.6B). If the area of marsh provides habitat and greater foraging opportunity for juvenile GuM when it is inundated, then the relationship seen here can be explained by the significant decrease in marsh area. Until 1985, edge continually increased, potentially allowing nekton increased access to marsh. However, since that peak in edge distance (Figure 3.10), edge has decreased concurrently with the decrease of marsh area. Considering factors together, marsh habitat (edge and internal marsh) may no longer provide the same historical benefits. Moreover, as the marsh degradation process occurs, marsh ponds with no direct connection with the estuary can become hostile environments for even the most highly adapted estuarine species. The results

of the multiple regression analysis and Chapter 2 may also support that notion that marsh area more than edge is an important factor in GuM production. That analysis showed that 61% of the variability in GuM abundance could be attributed to area of marsh and temperature ($R^2 = .61$, $p < .0001$). So while GuM may be thought more of a pelagic, open-water species, these findings suggest, that juveniles may find benefits from marsh habitat.

This study also found that assessing the relationship between BC landings and marsh habitat can lead to an ambiguous understanding of ecosystem dynamics. While it has been shown that vegetated habitats can reduce mortality of BC (Zimmerman et al. 2002), comparisons of landings data and landings CPUE data with marsh area indicates the opposite (Figure 3.9B-C). This study showed that BC landings and landings CPUE data increase with decreasing values of marsh edge, while survey data reveal trends consistent with previous findings, abundance decreasing with decreasing area of marsh (Figure 3.9A-C). The results of the BC survey data and marsh area regression are further supported by the results of the multiple regression analysis from Chapter 2, which found the area of marsh to be a significant factor in explaining this species' relative abundance ($R^2 = 0.33, p = 0.001$). Results of all three data types for BC on marsh edge distance show varied and contrasting trends. Generally, BC relative abundance remains constant over most values of marsh edge, while landings data shows a strong increase over increasing values of marsh edge, and landings CPUE data shows no particular trend. These varied results of the BC analysis highlights the importance of using fishery independent data when addressing habitat use questions for this species.

It's not surprising to find a strong relationship between BC abundance and the area of marsh, since this species often utilizes the marsh interior. However, one might suspect that BC may also show a strong relationship with marsh edge, but the regression of BC survey data on

marsh edge distance does not support this assumption (Figure 3.8D). Blue crab abundance does decrease over time (concurrently with decreases in marsh area) and this decrease occurs most rapidly between 1985 and 1995. However, the results of the tree regression for BC indicates a rebound after the 10 year lull in abundance (Figure 3.13). These results suggest that BC populations in Barataria Bay may have been indirectly susceptible to decreases in edge as the area of marsh declined past the 1985 apex of marsh edge. It appears however, that BC have adjusted to the marsh losses in the bay, but by what mechanisms remains unclear.

Results of RD landings regressions on habitat are contradictory in nature, further emphasizing the use of survey data in future analyses. Red drum sample size was small for the survey CPUE data within Barataria Bay, so extending these analyses to other basins known for having greater densities of RD may bring to light some of the incongruous results here. The current status of the RD stock is relatively unknown (due to complications of obtaining fishery dependent data on a stock under harvest restrictions), so there is a need to obtain more information about what is either currently or historically driving the variability in abundance of RD in the northern Gulf of Mexico (Powers 2012). Continuing to investigate the population dynamics of RD with fisheries independent data may be the key to obtaining a better understanding of this species.

It is unclear why the maximum value of edge observed in 1985, along with the subsequent reduction in marsh area and edge after that time, had no effect on SStrout. The tree regression for SStrout did show a significant decrease in abundance after 1990, but data prior to 1986 is unavailable, so determining if the decline in abundance is due to loss of marsh area or edge is difficult. Moreover, regression analyses with habitat (marsh area and edge) failed to show an effect on SStrout survey data; Chapter 2 failed to determine habitat variables as

important factors in explaining variability in SStrout abundance. Decreases in SStrout abundance can most likely be associated with fishing effort (see Chapter 2). Though the current stock assessment for SStrout indicated that overfishing is not currently occurring in Louisiana (West et al. 2011), patterns present in long-term fishery independent data reveal that continued increases in effort, coinciding with numerous other environmental insults in Barataria Bay, could pose potential threats to overall species abundance.

As the relative abundance for some nekton has remained stable in the face of marsh habitat loss, investigating potential changes in community structure may reveal other impacts caused by habitat loss. Further, ecosystems such as Barataria Bay that deal with multiple and long-term perturbations such as marsh habitat loss and fishing pressure, can experience what is known as the "shifting baseline syndrome" (Pauly 1995). In essence, fisheries scientists and managers enter the field of study and assess fish stocks and marine food webs and determine that the current state of affairs now describes how the ecosystem has always been. In most systems, historical differences in ecosystem structure and function cannot be detected because there are little data from the past (especially historical fishery independent data). In Louisiana however, we have a unique opportunity to use short-term historical data from the 1960's to assess how our ecosystem has changed (or not changed). Using non-parametric statistical approaches, results of the ANOSIM indicated that the three time periods compared were all significantly different from one another. However, it is the SIMPER analysis that provided insight as to the changes in SBDs between the time periods. Bay anchovy and AC are the most abundant species in Barataria Bay, so it follows that these species contribute to both the similarity of samples within each time period, but also are the most influential contributors to the time period pairwise differences. One of the most interesting findings becomes apparent when the similarities of

samples between each time are compared. The average similarity of Time 1 is the highest of all time periods at 33.71 and over 44% of the similarity between samples is attributable to BA. The story changes drastically in Time 3 when BA contribute to only 31% of the similarity between samples. In addition, the number of species which contributed to 90% of the global average similarity increases from eight species in Time 1 to ten species in Time 3. Most striking, however, is while BA and AC are still the most consistently caught species in samples in all time periods, their contribution to the samples decreased over time, giving way to other species. Considering these results, it appears that as the bay transitioned from a low marsh edge, high marsh area, low area of open water environment (Time 1) to a lower edge, lower marsh area and high area of open water environment (Time 3), the community structure became more diverse. The global average similarity of Time 1 and Time 3 also suggests that samples are becoming less similar, 33.71 and 27.49 respectively. The results of the similarities of samples between time periods also suggests that fishing pressure does in fact play an influential role in the variability of many estuarine nekton in Barataria Bay. For instance, in Time 2 (1990-1992), shrimping effort was still at relatively high levels, and had not yet succumb to some of the outside economic pressures that ultimately decreased fishing effort. The average abundance and thusly the percent contribution to similarity of BS, WS, and AC (common by-catch in the shrimp fleet), were all lower in Time 2 than in Time 3, when the effects of decreased fishing effort have already emerged. The higher abundances of penaeid shrimp and AC in Time 3 suggest that fishing pressure may be driving population dynamics, more so than habitat alterations. The results of this SBD analysis suggests that the baseline community in the late 1960's is different from the estuarine community of today. These changes are likely driven by fishing pressure, but to what degree habitat alteration has contributed to these changes remains to be determined.

This study highlighted some of the factors that may affect nekton abundance in estuaries. Knowing that these organisms exist in a dynamic environment, it will be important to investigate the influence of factors such as habitat change and changing salinity, along with the influence of predator-prey interactions. While the analyses in this chapter revealed patterns of abundance in both individual species and at the community level, looking more holistically at the system through ecosystem modeling is an important next step in understanding these complex relationships (Chesney et al. 2000).

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CHAPTER 4.
USING ECOPATH WITH ECOSIM TO ANALYZE THE RESPONSE OF ESTUARINE NEKTON ABUNDANCE TO CHANGES IN THE LINEAR DISTANCE OF MARSH EDGE

4.1 Introduction

Louisiana's coastal ecosystem has a long historical record of productive fisheries. Since the early 20th century, the coast has endured multiple perturbations resulting in many unforeseen ecological consequences (Day et al. 2007). One such consequence that is abundantly clear, however, is loss of marsh vegetation in southern Louisiana (LA). Marsh habitats are thought to play a role as nursery habitats for post-larval and juvenile fishes, providing both refuge from predation and foraging opportunities (Teal 1962, Gunter 1967, Nixon 1980, Boesch and Turner 1984, Houde and Rutherford 1993, Zimmerman et al. 2002). It should follow that with the reduction in habitat (levels of loss reaching as high as a football field an hour), fisheries production (yields) would decrease noticeably. Empirical evidence showing the direct link between estuarine nekton and saltmarsh habitat, however, remains elusive. Yet, other studies report that the link to saltmarsh habitat may not be as significant as once thought (Lee 2004, Fry 2008).

The leveeing of the Mississippi River (MR) interrupted the natural deltaic cycle in LA, a cycle that has been occurring for thousands of years. By its very nature, delta lobe formation and abandonment have long presented a changing marsh habitat to those species that occupy the estuaries. Therefore, the resilience seen in many estuarine nekton may simply be the result of those species adapting over time to deal with a constantly changing system. Though previous studies have not provided empirical evidence to support this hypothesis.

Past studies suggest that during the process of marsh degradation, the distance of marsh-edge interface (hereafter edge) will initially increase, thus providing increased marsh access to fishes that recruit to estuaries. After reaching a maximum, a steady decrease in edge length is expected, coinciding with a noticeable decrease in fisheries yields (Browder et al. 1989). This pattern of habitat degradation (Browder et al. 1989) is occurring in Barataria Bay and maximum edge length was achieved in 1985 (Figure 4.1). This short-lived increase in edge during marsh degradation is one mechanism that may explain the long-term stability in fish abundance observed in LA estuaries. Yet, it has been over 25 years since edge length reached its maximum. Since that time, the amount of edge has declined steadily in Barataria Bay, while commercial and recreational fisheries remain stable or increasing, calling to question Browder et al.'s (1989) conceptual model.

In Chapter 3, I concluded that species such as brown shrimp (BS, *Farfantepenaeus aztecus*), white shrimp (WS, *Litopenaeus setiferus*), and blue crab (BC, *Callinectes sapidus*) with previously reported affinities with marsh habitat (e.g. edge and area of intertidal marsh), had little change in relative abundance over the long term. Chapter 3 also included Gulf menhaden (GuM, *Brevoortia patronus)* and spotted seatrout (SStrout, *Cynoscion nebulosus*), whose overall relative abundances have decreased drastically in recent decades. The decrease in relative abundance for SStrout could partially be explained by changes in fishing effort. Gulf menhaden exhibited a significant relationship with marsh habitat, but it could not be determined if the decrease in abundance was marsh loss related, fishery related, or both. Given that the results in Chapter 3 have provided incongruous results with previously held hypotheses, in addition to the fact that fisheries landings have not seen a noticeable decline for some of the most important

commercial species, there is a need to explore this topic at the ecosystem level in an attempt to determine if other factors are driving variations in estuarine nekton abundance in coastal LA.

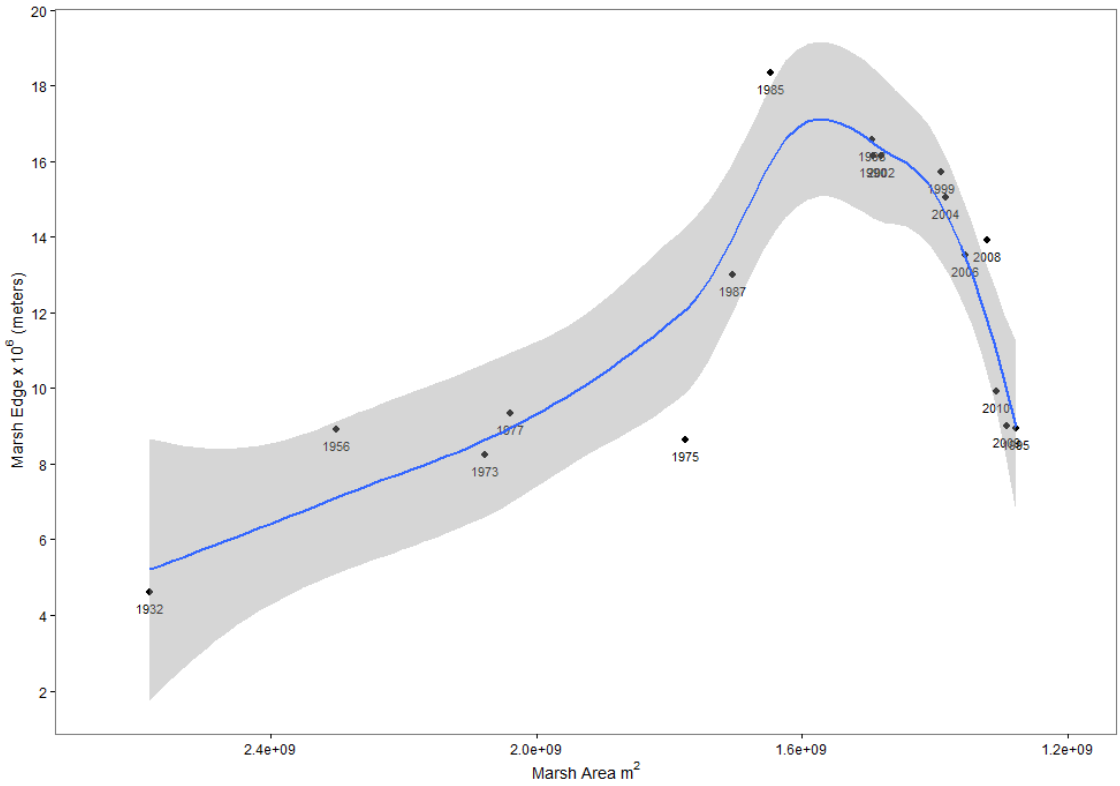

Figure 4.1. Plot of marsh area and the linear distance of marsh edge from Barataria Bay from 1967 to 2010. The maximum linear distance of marsh edge was achieved at ~1.9 x 10^4 km in 1990, all values of marsh edge after 1990 decreased from that apex (Couvillion et al. 2011).

To further evaluate the validity of Browders's (1989) conceptual model, I used Ecopath with Ecosim (hereafter EwE; Plaganyi 2007, Walters et al. 2009, Coll et al. 2009, Christensen et al. 2014) to determine the impact that edge had on nekton abundance (biomass) in Barataria Bay, LA. I tested the null hypothesis that marsh edge has no effect on nekton abundance in Barataria Bay. Recognizing that estuarine communities are subject to multiple sources of variation, this study considered a number of drivers within the ecosystem to determine if the model could closely hind cast historical abundances. To address this goal, I developed a mass-

balanced model that represents the Barataria Bay ecosystem as a snapshot in time (1967) based on long-term fisheries survey data collected by the Louisiana Department of Wildlife and Fisheries (LDWF). Then, to analyze the effects of multiple drivers on nekton abundance, the time-dynamic module Ecosim was first fit to historical relative biomass time series and salinity data and then forced with a time series of edge habitat data. Specific to LA, the driving forces within the system are confounded by the number of perturbations the system experiences. By including environmental drivers and accounting for trophic interactions, I shed more light on the long term effects of marsh loss on fish abundance in LA.

4.2 Methods

4.2.1 Study Area and Fishery Description

The LA coast is separated into seven Coastal Study Areas (CSAs) to delineate the natural geographic changes that occur from east to west along the coast. This study focuses on CSA 3, which encompasses Barataria Bay, a 6280 km^2 sub-region of the Barataria-Terrebonne estuary system and member of the National Estuary Program (Nelson et al. 2002). Barataria Bay (Figure 4.2), isolated from the MR (which is the easternmost boarder of the basin) since the 1940's, gets a majority its freshwater input from rainfall and includes freshwater, brackish, and marine coastal marshes. The basin is bounded on the west by Bayou Lafourche, an abandoned distributary of the MR (Conner and Day 1987). In addition to rainfall, the Davis Pond Freshwater Diversion provides a controlled flow (~28 m^3/s) of freshwater into the upper reaches of the estuary.

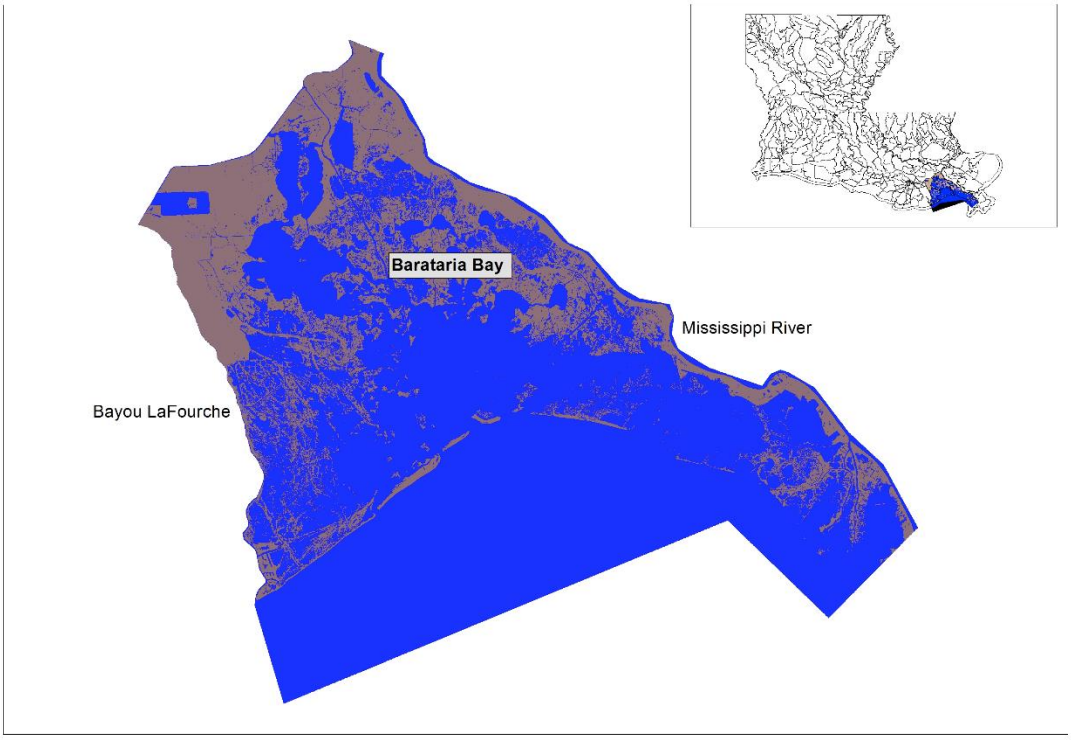

Figure 4.2. Barataria Bay, LA, USA, bordered on the east by the Mississippi River and on the west by Bayou LaFourche (Couvillion et al. 2011).

4.2.2 Nekton Species Included in EwE Model

There are a number of important commercial and recreational fisheries that occur in Barataria Bay. Spotted seatrout (*Cynoscion nebulosus*) and red drum (RD, *Sciaenops ocellatus*) comprise the majority of recreational catch within all LA estuaries. Commercial and recreational harvest of RD has changed significantly in the past 50 to 60 years. A majority of the harvest prior to the 1980s was commercial, with dramatic increase in catch concurrent with the rise in popularity of "blackened redfish" mid-decade. The increased effort led to a substantial decrease in spawning stock biomass, which led to development of a Fishery Management Plan (FMP), and a moratorium on commercial catch in federal waters in 1987 (Powers 2012). Recreational catch of RD also experienced a spike in landings in the mid-1980s and has steadily increased while

commercial landings remain at low levels. Brown shrimp, WS, GuM, and BC are all important commercial fisheries that occur in Barataria Bay.

Louisiana's shrimp fishery lands the most shrimp by weight compared to anywhere else in the US, and is second only to Texas for the most valuable shrimp fishery. Shrimp landings have experienced an obvious increase since the 1970's, even considering the reduction of fishing effort seen since around 2002 (as a result of outside market pressure from the Asian shrimp industry, a rise in fuel prices and a reduction in infrastructure after Hurricanes Katrina and Rita) (Minello and Zimmerman 1991, Zimmerman et al. 2002).

During the 20th century GuM were subject to an extensive purse seine fishery, with landings steadily increasing through the mid-1980s, followed by a decline. More recently, the landings have increased and stabilized in the last 10 years with no apparent trend. Louisiana alone harvests 92% of the annual GuM catch for the entire Gulf of Mexico fishery (Vaughan et al. 2007).

Commercial landings of BC have been recorded as far back as 1880; however little information regarding historical recreational landings has been found. There was an increase in commercial effort and landings between the 1980s and 1990s; more recently however, both effort and landings have stabilized (Guillory et al. 2001).

The two most abundant fish species in Barataria Bay are bay anchovy (BA; *Anchoa mitchilli*) and Atlantic croaker (AC; *Micropogonias undulates*). There are currently no active fisheries for these species, but they are by-caught in shrimp trawls.

4.2.3 Data collection and preparation

Data for the Ecopath base model, biomass time series (model calibration) and salinity time series (model driver) for use in Ecosim were collected by the LDWF Fishery Independent Shrimp/Finfish Monitoring Programs which began in 1966 and 1986 using 4.8 m trawls and

gillnets, respectively. For both trawls and gillnets, all sampling stations located within Barataria Bay were included in data calculations. For a detailed description of how fishery independent data were collected and processed for inclusion in the model see Chapter 2.

I used time series data of edge interface to drive the EwE model for Barataria Bay. For methods on how calculations for edge were determined, please refer to Chapter 2.

4.2.4 Ecopath Model: Structure and Assumptions

Ecopath with Ecosim is an open source ecosystem modeling software, originally developed by Polovina (1984) to model trophic interactions and to estimate mean annual biomass on a coral reef ecosystem. Since that time, the model has been greatly improved and is used in ecosystems worldwide (Christensen and Pauly 1992, Walters et al. 1997, Walters et al. 1999, Walters et al. 2000).

The first step in developing an EwE model is to create a well-described and documented mass-balanced model food-web model; EwE version 6.4 was used in this study (Christensen 2008). Two master equations must be satisfied to correctly parameterize the Ecopath model. The first equation describes the production of each functional group as a set of *n* linear equations for *n* groups:

$$\left(\frac{P_i}{B_i}\right) \cdot B_i \cdot EE_i - \sum_{j=1}^{n} B_j \cdot \left(\frac{Q_j}{B_j}\right) \cdot DC_{ji} - Y_i - E_i - BA_i = 0 \qquad \text{Eq. 4.1}$$

where $\left(\frac{P_i}{B_i}\right)$ is the production to biomass ratio for group i, EE_i is the ecotrophic efficiency (the proportion of production used in the system), B_i and B_j are the biomasses of the prey and predators respectively, $\left(\frac{Q_j}{B_j}\right)$ is the consumption to biomass ratio, DC_{ji} is the fraction of prey i in

predator j's diet, Y_i is catch rate for the fishery for group i, E_i is the net migration rate, and BA_i is the biomass accumulation for group i.

The underlying assumption of the Ecopath base model assumes that the conservation and transformation of energy through a system is conserved and the inability of system compartments to utilize all within the bounds of the modeled system. This idea can be simply stated below and is the second master equation for the initial mass-balanced model:

Consumption = production + respiration + unassimilated energy; Eq. 4.2

where production can be described as:

Production = predation mortality + catches + net migration + biomass accumulation + other mortality. Eq. 4.3

More succinctly, production can be described by the following equation

$$P_i = \sum_j Q_j \cdot DC_{ji} + (F_i + NM_i + BA_i + M0_i) \cdot B_i \quad \text{Eq. 4.4}$$

where P_i is the production of prey group i, $Q_j$is the consumption of predator j, DC_{ji} is the diet composition contribution of i to j's diet (by weight, not energy), F_i is the instantaneous rate of fishing mortality, NM_i is the net migration rate of prey group i, BA_i is the biomass accumulation rate for i, $M0_i$ is the other mortality rate for i (non-predation, non-fishery), and B_i is the biomass of i.

The Barataria Bay Ecopath Model (BBEM) represents what the food web in the bay might have looked like in 1967 with mean biomasses calculated from 1967-1969 data. The consumer groups considered in this study include zoobenthos, zooplankton, crustaceans, and fish. Values for zoobenthos (which includes the biomass of grass shrimp, *Palaemonetes sp.*) and

zooplankton were obtained from the Breton Sound EwE model (De Mutsert et al. 2012), while biomass values for all fish and crustaceans were utilized from local, Barataria Bay specific survey sample data. Functional groups for nekton were determined for inclusion in the model using preliminary analyses of species abundance from survey data. Some species known to inhabit Barataria Bay, but present in low numbers in sample data (e.g. red drum) were also included as functional groups because of their ecological importance in the system or importance as a recreational fishery. Louisiana Department of Wildlife and Fisheries utilizes different gear types to target certain species; using this a priori information I calculated the overall catch (combined over years) by gear type to determine which gear provided the highest catches per species. The species and species groups used in this analysis accounted for over 90% of the total catch in trawl gear samples (7 taxa), and over 18% of the total catch in gillnet samples (2 taxa) (Table 4.1). Producer groups (detritus, benthic algae, submerged aquatic vegetation (SAV), and phytoplankton) were also included in the BBEM and were borrowed from the Breton Sound EwE model (De Mutsert et al. 2012) (with the exception of SAV, which was estimated from personal observations made by K. de Mutsert).

Landings data for the recreational fishery (RD and SStrout) and landings data for the penaeid shrimp fisheries were also included as initial conditions in the model. Recreational landings were estimated at low values as those data were not available prior to 1981, while shrimp landings were derived from the NOAA Commercial Landings Statistics website following procedures form Chapter 3 of this volume (NOAA Fisheries 2013). Low levels of bycatch in the trawl fishery were also included in the model for BA and AC.

Table 4.1. Summary of total catch for the functional groups included in the Ecopath base model for years 1967-1969 for trawl and 1986-1990 for gillnet.

Functional Group	Gear	Catch	Total Catch	Percent Total Catch
Bay anchovy	trawl	137874	212088	65.0%
Atlantic croaker	trawl	24848	212088	11.7%
Brown shrimp	trawl	13554	212088	6.3%
Blue crab spp.	trawl	2522	212088	1.2%
white shrimp	trawl	2308	212088	1.0%
Gulf menhaden	trawl	10186	212088	7.0%
spotted seatrout	gillnet	3792	21837	17.3%
red drum	gillnet	283	21837	1.2%

Twenty-one functional groups (species or species guilds) that represent system and biomass dynamics, were defined. Major nekton groups were characterized as multi-age or multi-stanza groups to better represent the ontogenic changes through a species' life history (Christensen and Walters 2004). Each stanza in Ecopath requires input for the production to biomass ratio $\left(\frac{P_i}{B_i}\right)$, while biomass$(B_i)$, consumption to biomass ratio$\left(\frac{Q_j}{B_j}\right)$, and Von Bertalanffy K values (Von Bertalanffy 1933) are only required for the leading stanza allowing for Ecopath to estimate parameters that were not entered.

A diet matrix for each stanza group must be established within the base model and is typically one of the most challenging parts of model development. A consumer can feed proportionally on any prey item ranging from 0 to 1, with a predator's entire diet composition summing to a total of 1. The diet for each species was determined using information posted on fishbase (www.fishbase.org) and from published literature. The total biomass of any prey item within a system determines its availability for predation. Ecopath generates an ecotrophic

efficiency parameter (EE) that ensures a prey item is being consumed within realistic bounds of its overall biomass. If a stanza group's EE is > 1 during model parameterization, the stanza is most likely being over-consumed (either by predation mortality or fishing mortality) and the model will not balance. The diet matrix or landings data for that prey item will need to be iteratively adjusted until the model achieves mass-balance.

Relative biomass in the form of catch-per-unit-effort per m^2 ($CPUE/m^2$) was used for Ecopath biomass inputs to control for year-to-year variations in sample effort (De Mutsert et al 2012). Functional group biomasses were calculated by determining mean abundances for the years 1967-1969 and then converting the numbers of nekton to CPUE by weight using a length-weight (L-W) regression, $W_i = aTL_i^b$, where *a* and *b* are the species specific parameters, TL is length in total length, and *i* is each species of interest (Table 4.2). Because length data were unavailable for GuM, relative biomass for this species was calculated by determining the average weight of a 20 cm GuM (De Mutsert 2010). Then, using a L-W regression, an average weight of an adult GuM was determined and the product of this value and total catch over all years resulted in CPUE by weight.

For SStrout and RD, data were not collected via gillnets until 1986, so for the 1967-1969 estimates I used a 5 year mean from 1986-1990 for the baseline input values for these two species. All functional group CPUEs were then divided by the area swept by gear type to determine the $\frac{CPUE_i}{m^2}$, where i defines each functional group.

Table 4.2. Weight-Length regressions and data sources used to calculate CPUE in biomass. Red drum lengths (TL) were converted to FL using a Length-Length regression $FL = (L_{RD} + 2.394)\ x\ (0.916)$, as the only available parameters were evaluated at FL. **Blue crab were measured using carapace width.

Functional Group	Weight-Length Regression	Data Source	Units	
			Weight	Length
Spotted seatrout	$W_{sst} = 1.13\ x\ 10^{-8}\ TL_{sst}^{3.01}$	(Nieland et al. 2002)	kg	mm
Red drum*	$W_{rd} = 0.01FL_{bs}^{2.938}$	(Murphy and Taylor 1990)	g	cm
Brown shrimp	$W_{bs} = .006TL_{bs}^{2.938}$	(Fontaine and Neal 1971)	g	cm
White shrimp	$W_{ws} = .003TL_{ws}^{3.247}$	(Fontaine and Neal 1971)	g	cm
Blue crab**	$W_{bc} = .008CW_{bc}^{2.45}$	(West 2013)	g	mm
Bay anchovy	$W_{ba} = 0.0171TL_{ba}^{2.814}$	Fishbase	g	cm
Atlantic croaker	$W_{at} = 0.005TL_{at}^{3.148}$	Fishbase	g	cm
Gulf menhaden	$W_{gm} = .008TL_{bc}^{2.45}$	(De Mutsert 2010)	g	cm

3.2.5 Ecosystem Network Analysis

Ecosystem Network Analysis (ENA) is a method of examining ecosystem structure. More specifically, ENAs allow for comparisons with other ecosystems and has been incorporated into Ecopath as a way to measure the flow of energy and material between different compartments of an ecosystem. This analysis can also measure the efficiency by which energy is used, transferred and assimilated within the system (Baird 1993). To complete this analysis, trophic levels are aggregated into discrete levels rather than fractional levels, *sensu* Lindeman (1942), which allows for the calculation of numerous statistics (see below) that describe the system as a whole. Some basic summary statistics calculated for the system and included in this analysis were the sum of all respiratory flows (an important indicator for systems as this value tends to increase if the ecosystem is stressed), the sum of all production, net system production, total biomass (excluding detritus and assumed to increase as systems mature), and Total System

Throughput (the sum of all the flows that exist in a food web and refers to the overall activity of the ecosystem). Transfer Efficiencies (TE) are also calculated by the ENA and are described as the fraction of energy passed into a discrete trophic level that is then passed on to the next trophic level. Transfer efficiencies are generally known to be about 10% for many coastal systems (Lindeman 1942; Christensen and Pauly 1993). There are also a number of values output as ratios that are helpful in describing different characteristics of the ecosystem, these include: Total primary production over total biomass (developing systems tend to have a high P/B because they are characterized by low biomasses and high production), and total primary production over total respiration (this value will increase with increased maturity of the ecosystem) (Odum 1971). The ENA was used to compare the baseline BBEM to other models from similar systems to determine if the model metrics calculated by the BEEM were within realistic bounds.

4.2.6 Ecosim Model: Structure and Assumptions

Applying the initial parameters derived from the first master equation in Ecopath, the Ecosim module of EwE can be invoked. Ecosim re-expresses the system of linear equations from Ecopath as a system of coupled differential equations to predict future outcomes (Eq.5):

$$\frac{dB_i}{dt} = g_i \sum_j C_{ji} - \sum_j C_{ij} + I_i - (M_i + F_i + e_i)B_i \,, \qquad \text{Eq. 4.5}$$

where g_i is the net growth efficiency; I_i is the biomass immigration rate; M_i is the nonpredation mortality rate; F_i is the fishing mortality rate; e_i is the emigration rate; C_{ij} can be considered the "flow" from pool i to pool j organisms as a function of time—the consumption rate of type i biomass pool by type j biomass pool. In Ecosim, the rates of consumption can be limited at very small temporal and spatial scales, allowing for the flow of prey (v_{ij}) from (behaviorally or

locationally) varying states of vulnerability to limit the rates of predation to levels that the traditional Lotka-Volterra mass-action models would not predict (Walters et al. 2008). Consumption rates in Ecosim can be described by:

$$Q_{ij}(B_i, B_j) = \frac{a_{ij} v_{ij} B_i B_j}{(v'_{ij} + v_{ij} + a_{ij} B_j)} \quad , \qquad \text{Eq. 4.6}$$

where B_i is the biomass of the prey; B_j is the biomass of the predator/consumer; a_{ij} is the rate of effective search for prey i by predator j; v_{ij} and v'_{ij} are the behavioral exchange rates between prey pools, expressed as vulnerable and invulnerable. Equation 4.6 is based on the concept of the foraging arena theory, which regulates consumption rates by assuming predator-prey interactions take place in restricted arenas where prey vulnerability in terms of predation depends on a prey's need for a particular resource. (Walters et al. 1997, Ma et al. 2010). Vulnerability in predator/prey interactions can also be influenced in Ecosim by the addition of meditation factors, M_{ij}, which allows for a third organism to affect a predator/prey pair's interaction. Further, environmental factors can also influence trophic interactions by including forcing functions, S_{ij} , which are used to alter the effective search rate a_{ij}:

$$Q_{ij}(B_i, B_j) = \frac{a_{ij} v_{ij} B_i B_j T_i T_j S_{ij} \frac{M_{ij}}{D_j}}{v'_{ij} + v_{ij} T_i M_{ij} + a_{ij} M_{ij} S_{ij} B_j \frac{T_j}{D_j}} \qquad \text{Eq. 4.7}$$

where T_i is the relative feeding time for prey, T_j is the relative feeding time for predators, and $D_j$is the effect of handling time to limit the rate of consumption (Christensen and Walters 2004). The transfer rate v_{ij} values determine whether control within the ecosystem is top-down or bottom-up, where high values indicate top-down control and low values indicate bottom-up control (Christensen and Walters 2004).

4.2.7 Calibration

An important step in using and applying dynamic ecosystem models is to ensure they can reproduce historical patterns of abundance for further use in policy analysis and future predictions (Shannon et al. 2004). Catch-per-unit-effort data for eight nekton functional groups derived from LDWF surveys were used to calculate annual relative biomass time series for the Ecosim calibration procedure.

To calibrate the models used for this study, annual relative biomass data, annual edge data, and annual mean salinity data were used in the Fit-to-Time Series module of Ecosim. This module was used to find predator-prey interactions that were most sensitive to changes in v_{ij} from the nekton groups for which time series data were available. The model then estimates v_{ij} values that produce a better fit to the observed data. To assess the fit of the model, the sum of squared deviations (SS) of the observed logarithmic (log) biomass values was used (Christensen et al. 2005). Salinity forcing data were applied using procedures from De Mutsert et al. (2012). A model with salinity and relative biomass time series data was fit and a baseline Sum of Squares (SS_i) was recorded.

After the baseline model was determined, three different amplitudes of edge data were tested (Figure 4.3). These scenarios were used to determine which configuration explained the most variability in the time series data and were developed using the following transformations:

$$Edge_i = stdEdge^{1.0} \quad \text{Eq. 4.8}$$

$$Edge_d = stdEdge^{0.2} \quad \text{Eq. 4.9}$$

$$Edge_e = stdEdge^{3.0} \quad \text{Eq. 4.10.}$$

$Edge_i$ are the initial marsh edge values calculated by standardizing each time step to the long term mean value, $Edge_d$ are the dampened marsh edge values calculated by applying a power function with exponent 0.2 to the standardized marsh edge values ($stdEdge$), and $Edge_e$ are the expanded marsh edge values calculated by applying a power function with exponent 3.0 to the standardized marsh edge values ($stdEdge$). Specifically, each edge scenario ($Edge_i$, $Edge_d$, $Edge_e$) was fit separately using the fitting algorithm described below. The best fitting models were chosen by assessing both the change is SS from the baseline model (SS_i) and the Akaike Information Criteria (AIC) value that penalizes models with more parameters. Smaller values of AIC indicate a better fit.

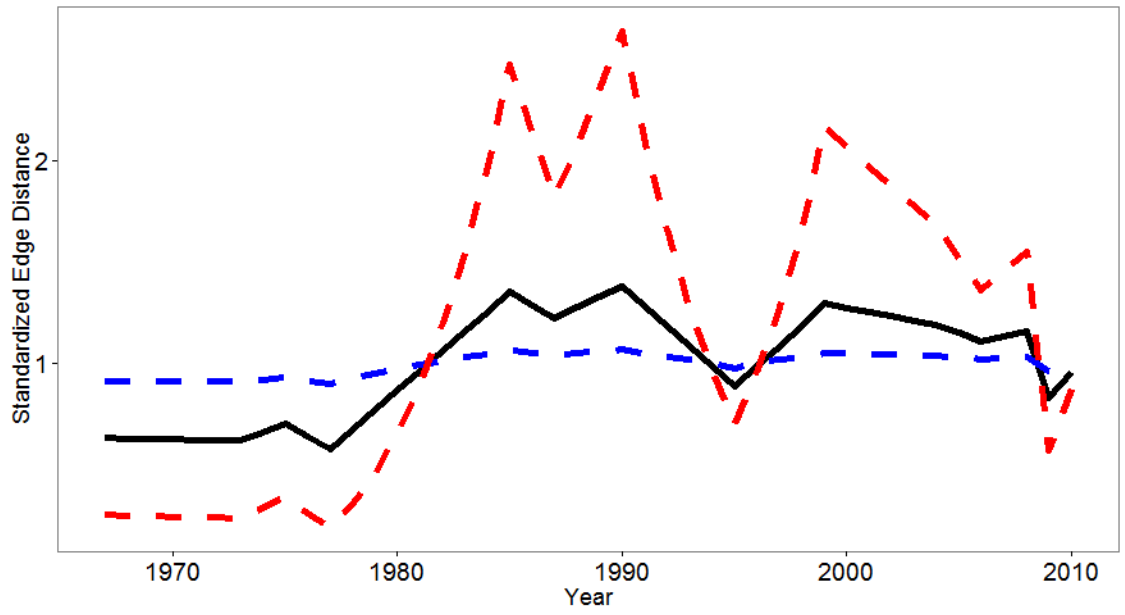

Figure 4.3. Marsh edge time series power functions used in preliminary runs in the Ecosim model. The black line indicates the standardized marsh edge values, the red dashed line indicates the standardized marsh edge values when raised to a power of three, and the blue dashed line indications the standardized marsh edge values raised to a power of 0.2.

Marsh edge forcing can be applied to all or a chosen number of predator-prey interactions, while also choosing whether to alter the search rate, vulnerabilities, the foraging

arena area, or the vulnerability and the arena area concurrently. In addition, those species known to occupy the marsh when inundated, forcing functions were also applied to those taxa. The edge forcing function in this study was applied to the effective rate of search (a_{ij}).

The following procedures were used to calibrate the 1967-2010 BBEM, modified from methods of Shannon (2004), Christensen and Walters (2008), Coll et al. (2008), and Howell et al. (2012):

1) *Balanced model check*. Prior model fitting and tuning, an Ecosim run was invoked with no forcing data or time series data to ensure a balanced model from Ecopath was being utilized (Christensen et al. 2008).
2) *Data input*. All time-series, salinity, and edge forcing data were read into the EwE model database. Salinity and edge data were used to drive the model and relative biomass data were used to assess the fit of the model by comparing the observed data with the model predicted biomasses.
3) *Baseline model fit*. The relative biomass data were invoked and a baseline goodness-of-fit statistic (SS_i) was calculated in Ecosim following procedures in Christensen and Walters (2004).
4) *Vulnerability search*. The time series data for relative biomass and salinity were introduced to the model. The first edge forcing scenario was activated and the fit-to-time-series module was applied by using Search Groups with Time Series function to search for the values of v_{ij} that would minimize the SS from the SS_i. The v_{ij}values were then estimated by the model for the predator-prey interactions that were deemed most sensitive. This step repeated for each of the remaining marsh edge distance scenarios.

5) *Evaluation of model outcomes.* The three marsh edge scenario model runs were analyzed by comparing the observed and predicted biomass time series data and assessing the decrease in SS.

4.2.8 Sensitivity Analysis

The Monte Carlo (MC) routine provided by EwE can be used to test the sensitivity of Ecosim's outputs to changes in the initial Ecopath inputs. The user is able to set the number of trials where each trial uses randomly selected values for initial input parameters ($B_i, \frac{P}{B}, EE, BA$). These parameters are selected from a uniform distribution centered on the initial Ecopath input with a user-defined coefficient of variation (CV, defined here as the default 0.1). Essentially, this routine is looking for a set of Ecopath inputs, within the defined CV that would further minimize the SS for the model (Christensen et al. 2008).

4.3 Results

4.3.1. Ecopath and Ecosystem Network Analysis

A balanced Ecopath model of 1967 Barataria Bay was achieved iteratively by first adjusting the diet matrix, as diet compositions represent only snapshots of the feeding habits of individual species and are likely to be relatively variable based on location and time periods of data collection. Other input parameters were also adjusted iteratively, and when a balanced model was achieved, parameters were cross-referenced with other published Ecopath models to ensure the plausibility of each value. During the balancing procedure, EwE generated missing parameter values (Table 4.3).

Table 4.4. The biomass pools included in the Barataria Bay Ecopath model along with the basic inputs as required by Ecopath. Optimum and upper and lower standard deviation (SD) salinity values were also included, and we applied during the Ecosim runs. Data sources are indicated by superscript: (*)-K. de Mutsert, personal communication; ([1]) - data derived from Barataria Bay survey data; ([2]) - parameter estimated by EwE software; ([3]) – data obtained from the Breton Sound EwE Model (De Mutsert et al. 2012); ([4]) – data obtained from the Gulf of Mexico EwE (Walters et al. 2008); ([5]) – data obtained from the Weeks Bay EwE model (Althauser 2003); ([6])—data obtained from (Gandy et al. 2011).

Group Name	Biomass (g/m^2)	P/B	Q/B	VBGF K	EE[2]	Opt. Salinity[3]	Low. SD[3]	Upper SD[3]
juve. spotted seatrout	0.0002[2]	3.70[3]	6.349[2]	0.3	0.01	15.60	8.30	8.30
spotted seatrout	0.0039[1]	0.70[3]	1.6[4]	-	0.75	14.60	9.30	9.30
juve. red drum	0.000003[2]	2.30[3]	6.026[2]	-	0.02	26.50	13.50	13.50
adult red drum	0.00008[1]	0.62[3]	1.86[4]	0.4	0.62	9.20	6.80	6.80
juve. croaker	0.087[1]	2.0[3]	20.03[2]	0.75	0.64	17.00	7.00	7.00
Atlantic croaker	0.29[2]	1.50[4]	10[4]	-	0.23	17.00	7.00	7.00
juve. blue crab	0.043[2]	3.0[3]	16.77[2]	-	0.23	11.50[6]	6.40[6]	15.00[6]
blue crab	0.052[1]	2.40[4]	8.5[4]	0.75	0.70	7.60[6]	9.30[6]	20.00[6]
juve. white shrimp	0.010[2]	3.0[3]	45.16[2]	-	0.23	17.00	6.60	6.60
white shrimp	0.02[1]	2.40[4]	19.2[2]	0.3	0.89	9.80	6.60	6.60
juve. brown shrimp	0.033[2]	3.0[3]	33.43[2]	-	0.14	17.00	6.60	6.60
brown shrimp	0.067[1]	2.40[4]	14.21[4]	0.3	0.94	9.80	6.60	6.60
juve. menhaden	0.0019[1]	2.30[4]	11.54[2]	-	0.66	14.20	6.40	6.40
gulf menhaden	0.0038[2]	1.90[4]	6[4]	0.8	0.15	15.20	5.40	5.40
bay anchovy	0.10[1]	2.53[4]	14[4]	-	0.92	15.00	8.00	8.00
zooplankton	0.61[5]	28.77[5]	84.87[5]	-	0.03	35.00	1000	1000
zoobenthos	3.22[5]	4.50[3]	22[4]	-	0.32	35.00	1000	1000
phytoplankton	1.946[5]	101.70[5]	-	-	0.24	35.00	1000	1000
SAV	5.24*	9.01[4]	-	-	0.36	35.00	1000	1000
benthic algae	12.876[5]	3.91[5]	-	-	0.63	35.00	1000	1000
detritus	4.0[5]	-	-	-	0.12	-	-	-

A summary of trophic flows from the Ecopath base model shows the highest trophic level in the model to be 3.27 (adult RD) and the average trophic level of all nekton to be 2.57 (Figure 4.5).

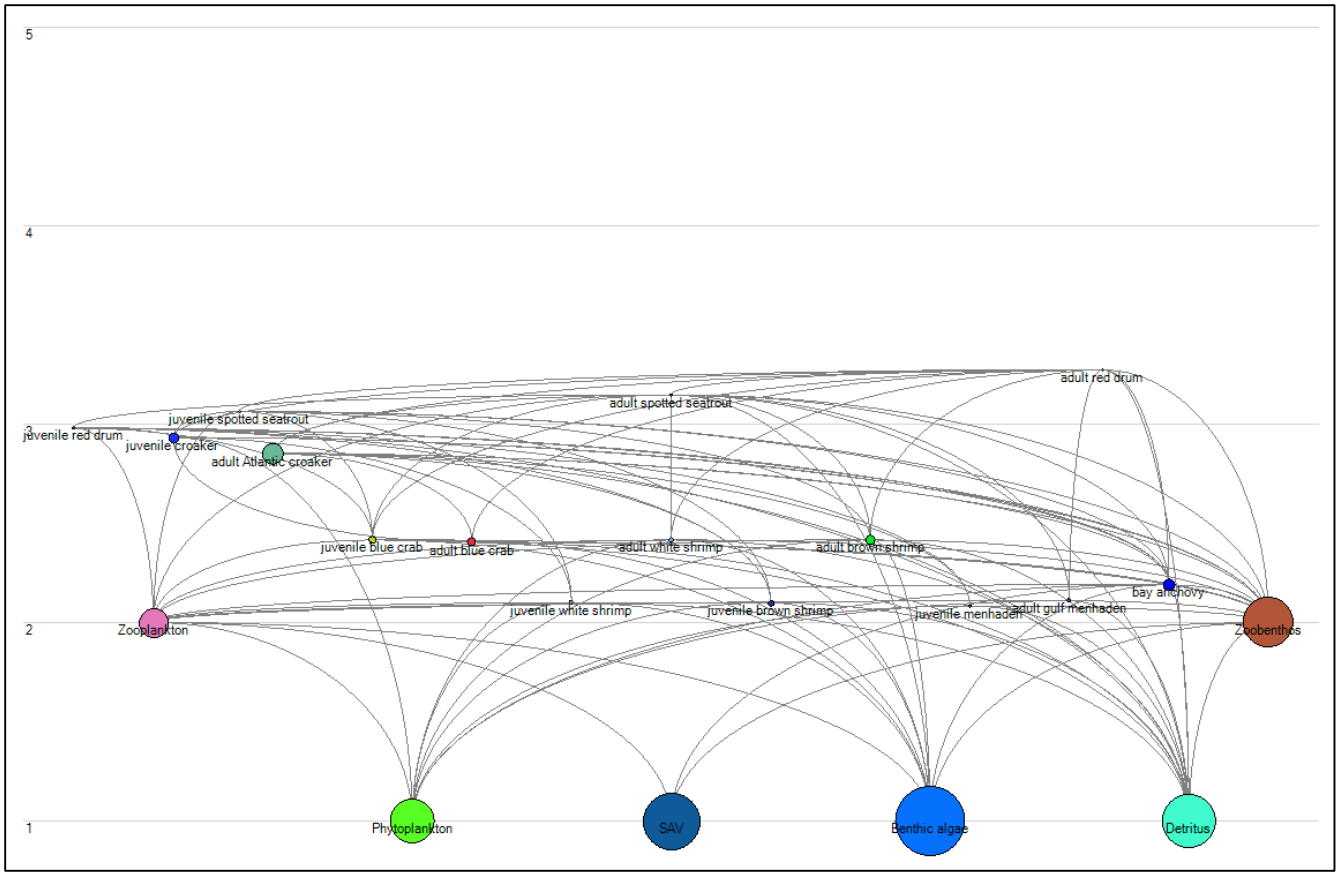

Figure 4.5. Flow diagram generated from the Ecopath base model for Barataria Bay in 1967. The size of the circles refers directly to the size of the biomass pools with their respective trophic levels indicated on the y-axis of the diagram.

3.5. Ecosystem Network Analysis

Ecosystem metrics were estimated for the base model in 1967 (Table 4.6). Compared to other systems, the BBEM model generally reports lower values for most of the ENA variables. Transfer efficiencies for the BBEM model were within the acceptable range, 5.3% and 7.2% for primary producers and detritus respectively (Christensen 2008).

Table 4.6. Comparison of ecosystem structure and function of the 2 future scenarios with the Barataria Bay base model from 1967 using the following factors: Sum of Respiratory flows (∑ of R flows), Total system Throughput (Total system T), Sum of all Production (∑ of all P), Total Primary productivity over Total Respiration (Total PP/total R), Net system Production (Net system P), Total Primary productivity over total Biomass (Total PP/total B), andTotal Biomass excluding detritus (Total B) (Global TE).

Parameter	Barataria Bay	Breton Sound	Terminos Lagoon	Laguna Alvarado	Units
∑ of R flows	72.6	130.9	-	987.5	$g\ m^2\ yr^{-1}$
Total system T	682.3	1730	3709.5	2683	$g\ m^2\ yr^{-1}$
∑ of all P	328.9	839.9	-	1574	$g\ m^2\ yr^{-1}$
Total PP/Total R	4.06	6.04	-	1.3	-
Net system P	222.8	660.3	2611.6	303.5	$g\ m^2\ yr^{-1}$
Total PP/Total B	12.01	8.72	-	16.5	-
Total B	24.6	90.8	-	78.1	$g\ m^2$

4.3.2. Ecosim, Calibration and Sensitivity Analysis

Using the model calibration procedures, a total of three different models were fit to salinity, relative biomass time series data and the amplitude of response marsh edge forcing scenarios. Model *SS* and values of AIC were used to assess the fit of each of the models (Table 4.5). In an attempt to address the potential varied response of taxa to environmental impacts, power functions were used to expand and contract the amplitude of response of the linear distance of marsh edge time series. Model fits were best using standardized raw data values or the dampened values generated by applying a power function of 0.2 to the data. While the model using the dampened effect of marsh edge forcing showed a slightly greater decrease in *SS* and a slightly lower AIC value, the difference was not statistically significant and it was determined that using the dampened marsh edge values would likely introduce more uncertainty into the model. The model chosen for use in future scenarios and all other comparative analyses was able to correctly reproduce 23% of the variability in the times series data.

Vulnerabilities were estimated by the Fit to Time Series module for the 8 taxa for which time series data were available. Any vulnerabilities that were >10 were iteratively reduced down

to a value of 10, with the model fit (SS) being investigated at each step to ensure no large changes in fit occurred during the adjustment (Howell et al. 2012) .

The MC routine successfully created a balanced model for each of the 20 trials by varying the input parameters within 10% confidence intervals. The SS for each trial model varied between 149.75 and 227.87.

Table 4.5. Results from the calibration procedure for the BEEM. The amplitude of response describes each power function that heighted or damned the marsh edge forcing time series. The models were fit iteratively with the SS and AIC values indicating the fit of each step in the model fitting process. The percent contribution is the variability accounted for by each step in the fitting procedure.

Amplitude of Response	**Model Run**	**SS Values**	**AIC**	**% Contri.**
stdEdge	First Run (Time Series)	198.4	-	-
	Salinity + Time Series	192.4	-	-3%
	Edge forcing	151.95	96.38	-20%
Edge $^{0.2}$	First Run (Time Series)	198.4	-	-
	Salinity + Time Series	192.4	-	-3%
	Edge forcing	151.1	96.28	-21%
Edge^3	First Run (Time Series)	198.4	-	-
	Salinity + Time Series	192.4	-	-3%
	Edge forcing	164.5	97.64	-14%

The BEEM was fit to time series of SStrout, RD, AC, BS, WS, BC, GuM and BA. The BEEM was driven by salinity time series data and marsh edge data. The model best captured the observed trends for SStrout, GuM, BA and WS, while it generally underestimated biomasses for RD. The model predicted biomasses for BS, BC and AC were incongruous to observed trends (Figure 4.5).

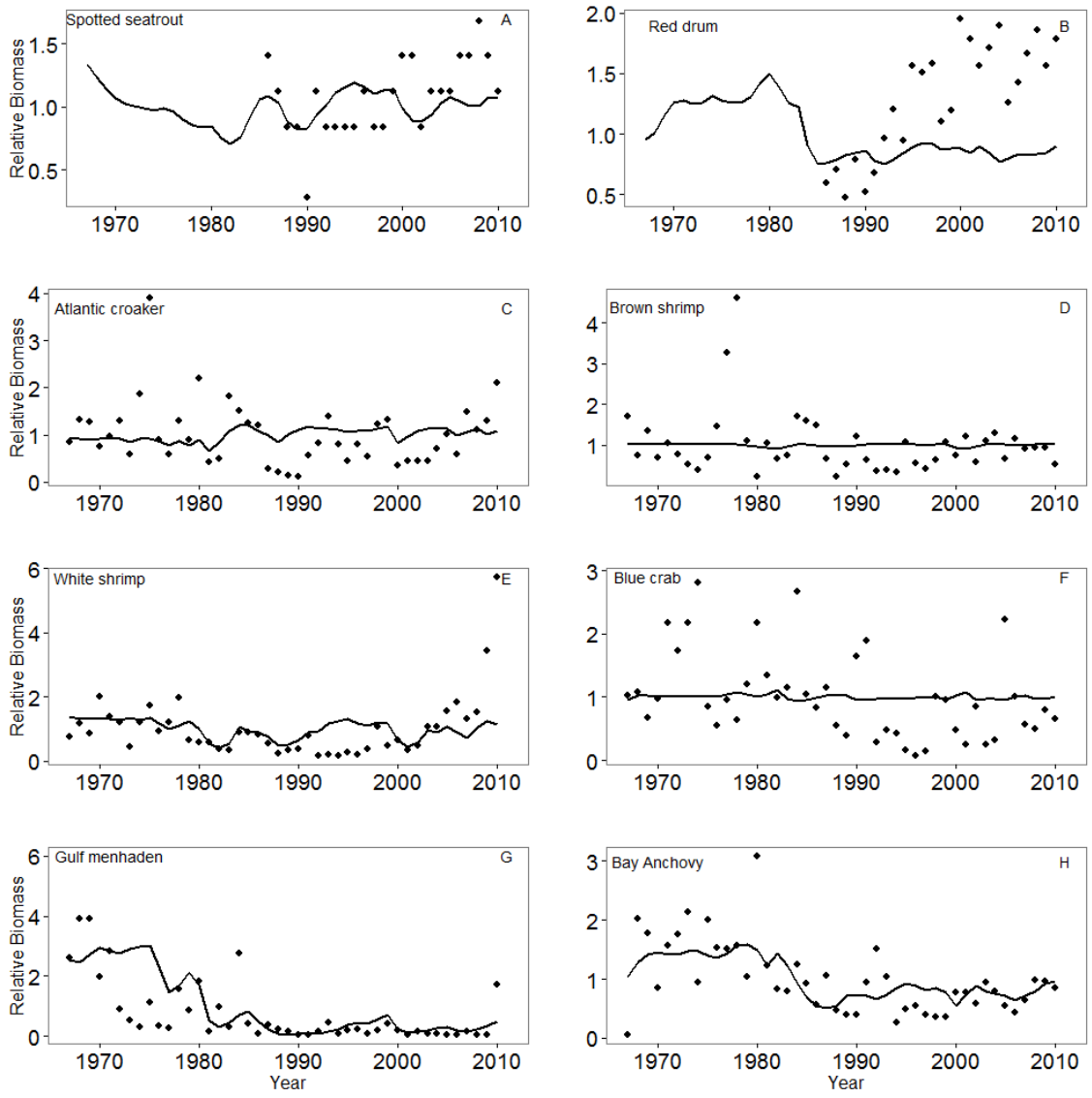

Figure 4.5. Ecosim model fit of the BEEM to LDWF relative survey data. The points indicate the relative survey data used to calibrate the model and the line indicates the simulated biomass of the functional groups for which survey data existed.

4.4 Discussion

This study examined the combined effects of changes in edge and salinity on trophic interactions and found that edge accounted for 20% of the variation in long-term abundance of nekton species. This result corroborates findings in Chapters 2 and 3, showing that marsh habitat (e.g. edge and marsh area) may be less important in driving estuarine nekton abundance than

once thought. Recent studies infer that marshes are likely to play a role in supporting fish production (Brown et al. 2013), but the magnitude of the support is difficult to estimate. The lack of a strong relationship between nekton species and marsh habitat, may not be surprising if we consider this phenomenon on a broader scale. Coastal LA is has been experiencing changes in the amount and extent of coastal marshes for thousands of years through the formation and abandonment of delta lobes. Species inhabiting this region may have evolved certain physiological and behavioral mechanisms that allow for adaption to changes in their environment. While the deltaic cycle has since been disrupted by the leveeing of the MR, change has still been occurring, both by anthropogenic and natural sources (e.g. storm events). It may be the very nature of these dynamic systems that allow for species to adapt to changes, even if the rate at which the changes are occurring has increased. If LA or the western Gulf is a unique by the very nature of it being a delta system, then the direction and focus of future research on the resilience of estuarine nekton can be better guided. Therefore, determining if marsh loss in non-deltaic saltmarshes are affecting estuarine nekton differently (i.e. are abundances decreasing concurrently with changes to marsh habitat?) may be the next step in disentangling this issue.

Although this EwE model did not include an exhaustive list of taxa as functional groups, the most important taxa in the ecosystem were included. Earlier versions of this model included 40+ functions groups and showed similar outcomes as the more parsimonious model used here with only 21 functional groups.

Fitting the Ecosim model to relative-biomass time series data resulted in a better fit to historical data. In terms of overall fit to the data, using only 8 relative-biomass time series, along with salinity and marsh edge forcing data, the model was able to account for 23% of the variation in the time series data. Considering the known variability in biological systems, a 23% reduction

in the SS is actually quite notable. However, the BBEM was unsuccessful in accounting for a majority of the inter-annual variation seen in the observed data, and that result was not surprising. Results in Chapter 2 indicate that variability in species biomass can be attributed in part to landings data (SStrout). Fishing pressure in the form of time series effort data or stock assessment estimated fishing mortality data were deliberately excluded from the model to gain a better understanding of how changes in edge effected long term abundance of estuarine nekton. Fishing drivers were also omitted from the Ecosim model because commercial shrimp effort and fishing mortality data were both highly correlated with the time series of edge interface. While this correlation might draw attention and suggest these factors are indeed related, correlation, in this instance does not imply causation. The similarity in the long term trends of fishing data and marsh interface data is a matter of chance. The decreases in fishing effort (which mirrors the decrease in marsh edge) over the past 25 years can be better explained by economic factors such as increases in fuel prices and the import of Asian shrimp. Moving forward, constructing models that account for fishing pressure within the ecosystem will facilitate better fits to historical data, and can more confidently be used as a management tool for evaluating the relative effects of different restoration scenarios.

When using environmental data to inform a food web model, determining the amplitude of response for each taxa can be challenging, especially for complex food webs. Organisms have the ability to adapt to changes in their environment in ways that are not easily quantified, especially those species that have adapted to rapidly changing ecosystems. Species that spend at least part of their life history near or in estuaries have evolved physiological and behavioral adaptations that help them cope with the multiple stressors and changes that can occur over both short and long term time scales. Fitting three marsh edge scenarios allowed me to analyze three

potential amplitudes of response possible in this ecosystem. This approach by no means represents all responses of all the organisms to environmental variables, but it gave an idea of how sensitive nekton were to this particular driver. Considering the fits of the standardized raw and dampened marsh edge time series scenarios, one could conclude that because the standardized raw values produced a similar fit as the dampened values, the effect of edge on estuarine nekton abundance was limited at best. Moreover, even when I increased the amplitude of response by a power of 3, model predictions for the species previously shown to exist in higher densities near the marsh edge did not respond to that increase. The results of the Ecosim run for BS and BC, for instance, demonstrated that even when accounting for changes in salinity and trophic interactions, the model was unable to reproduce the variability seen in the time series data (Figure 4.5D). Further, Chapter 3 showed that BS and BC abundances are constant over most values of marsh edge, which is consistent with model predictions in this chapter. While many other studies have shown that BS and BC abundance is higher at the marsh interface, this study, among others, suggests that density at the marsh edge only tells part of the story of the variability in nekton abundance in estuaries (Lee 2004, Fry 2008).

Ecosystem models are always an over-simplification of the actual food web meant to summarize the important components of the system. With this understanding in mind, results of the ENA can help determine if the model in use is a possible working ecosystem with metrics similar to other models (De Mutsert et al 2012). Results of the ENA showed that, while most metrics were lower when compared to other systems, the BBEM is one possible model for this system. The ratio of total production/total respiration, was a slightly elevated at 4.06, as other models report values between 0.8 and 3.2 (Christensen and Pauly 1993). Often, Ecopath models tend to overestimate these values due to exclusion of bacterial activity in the food web, as was

the case in the BBEM. Bacteria utilize detritus within the system and then respire. Omitting this functional group can cause the model to underestimate total respiration in the system (resulting in an inflated production to respiration ratio), in addition to underestimating total system throughput. Transfer efficiencies reported by the ENA are also lower when compared to other systems, but a wide range of TEs have been previously reported and thus the values herein appear acceptable (Christensen and Pauly 1993). The sum of all production is similar to past estimates of this metric estimated at 360 g m^2 yr^{-1} (Houde and Rutherford 1993).

Effects of other environmental variables not included in this model could also be influencing the model predicted biomasses. Temperature was not included in the BEEM as the average temperature of Barataria Bay has not significantly changed over time. Turbidity, dissolved oxygen and water depth are also variables to consider that likely influence the abundance of nekton in the bay. However, by aggregating these variables annually, the potential effects on nekton are typically lost. Ecosim can use time-varying monthly inputs for environmental variables, so a next step for this model would be to obtain environmental drivers as monthly input and examine the results over the long term. However, I consider it unlikely that these excluded variables are more influential than external drivers, such as fishing pressure, because nekton typically move toward areas with favorable environmental conditions.

Previous studies concluded that increased productivity of penaeid shrimp and blue crab (in the form of fisheries yields) is linked to their utilization of saltmarshes (Minello and Rozas 2002, Zimmerman et al. 2002, Minello et al 2012). The authors of the aforementioned studies suggested that the edge effect described by Browder et al. (1985, 1989) explains the increase in fisheries landings in light of the marsh loss occurring in the Northern Gulf of Mexico. Indeed, if that model is at work in LA ecosystems, once the estuary becomes more than 50% open water,

and marsh edge begins to decline precipitously, then one could predict that fisheries yields would also decline. However, the maximum value of marsh edge was achieved in Barataria Bay in 1985, almost 30 years ago, and there has not been a noticeable decline in landings since that time for penaeid shrimp. Results from Chapter 3 also suggest that abundance of many species included in the EwE model have shown to be resilient over most values of edge and most values of marsh area. While I do not discount studies that showed higher densities of shrimp and other nekton nearer to marsh edge, it seems likely that there are other, much stronger drivers at work within this ecosystem.

It is clear that the relationship between saltmarshes and estuarine nekton is a complicated one, but results of this study suggest that the answer may not be fully explained by the edge effect as proposed by Browder and others (Browder et al. 1989). However, this study only represents one basin in coastal LA. To determine how robust these findings are, analyses should be extended to other coastal estuaries. Louisiana's coastal basins vary in both salinity gradients and the amount and rate of marsh loss. An interesting next step would be to conduct these same analyses in the Atchafalaya Delta where land is actually accreting, giving a proof of concept to the influence of saltmarsh habitat on fish abundance and production. In addition, research is underway to apply the spatial module of the EwE package, Ecospace, which will allow for nekton abundance to adjust in both space and time to internal and external drivers within the system. New developments in this module allows direct incorporation of GIS data, making it possible to simulate multiple responses of nekton to changing marsh habitat (Steenbeek et al. 2013). While the relationship between estuarine species and marsh habitat is still elusive, this study can be used to guide future inquiry.

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CHAPTER 5.
GENERAL CONCLUSIONS

5.1. Introduction

This study set out to determine what influence, if any, marsh loss had on the historical abundance of estuarine nekton in Barataria Bay, Louisiana (LA). Because marsh loss is occurring at high rates, coupled with the hypothesized importance of marsh habitat to the success of many estuarine species, understanding the mechanisms that drive variation in abundance is significant. This study also explored the differences between fishery independent and fishery dependent data as applied as indices of relative abundance. Previous studies used landings data to evaluate the influence of habitat change, under the assumption that landings are a reasonable indicator of relative abundance. To that end, this study sought to answer the following questions:

1. Are landings data an appropriate indicator of relative abundance for species of interest in Barataria Bay?
2. Is marsh habitat (area of marsh and distance of marsh edge interface) a significant driver of historical nekton abundance?
3. Did the distance of marsh edge interface increase fish production (edge effect), temporarily masking negative long term impacts of habitat loss on the abundance of estuarine species?
4. Has nekton community structure changed over the long term in light of the increase and subsequent decrease in marsh edge over the period of record?

5.2. Empirical Findings

In Chapter 2, I found that for most species landings and survey data show differing patterns of abundance through time and in Chapter 3 I found that landings and survey data show

differing patterns of abundance over most values of marsh area and marsh edge. Therefore using landings data as an index of abundance or to describe abundance patterns relative to habitat loss leads to relationships that may not be representative of nature. All three chapters revealed that for important nekton species such as penaeid shrimp and blue crab (*Callinectes sapidus*) previously thought to be strongly linked to marsh edge, these relationships appear weak, if not absent, even when accounting for other drivers such as salinity, rainfall, river input and trophic interactions. Chapter 3 results indicated the maximum value of marsh edge was achieved in 1985, so if the edge effect is indeed driving nekton abundance, then abundance of estuarine species should have declined shortly thereafter. For those species historically thought to benefit from marsh habitat and the edge interface in particular, a brief dip in relative abundance after 1985 was observed, but all three species (penaeid shrimp and blue crab) showed a rebound in abundance to pre-marsh edge maximum values and above. Though marsh related species abundance appear resilient in this study, changes in community structure could be an outcome missed by single species or even multispecies analyses. Results from Chapter 3 indicated that comparisons of species biomass distributions (SBD) from before, during, and after the maximum value of edge was achieved were significantly different. It is not clear, however, to what degree habitat loss has contributed to these changes as fishing has been occurring in the system over the same period of record. Finally, Chapter 4 indicated that after accounting for trophic interactions and salinity drivers, edge does not account for a significant portion of historical variation nekton abundance.

5.3. Theoretical Implications

Results of this study suggest that previous hypotheses explaining the relationship between marsh habitat and fish abundance may no longer be applicable. For instance, a number of studies (Turner 1977, Pauly and Ingles 1999) correlated the area of marsh with penaeid shrimp

yields, assuming that landings are a proxy for penaeid shrimp abundance. In Chapter 3, separate comparisons for penaeid shrimp species along with separate comparisons of marsh area with survey and landings data indicated that in Barataria Bay survey data showed different patterns abundance with compared with landings and landings catch-per-unit-effort (CPUE) data. The differences between these two analyses could be attributed to the analysis being aggregated by both spatial extent and by species. Previous studies have reported that using landings as an index for abundance is suitable for species such as penaeid shrimp and gulf menhaden because their life history essential makes them a "yearly crop." However, the findings here suggest that using landings to describe abundances of estuarine nekton in Barataria Bay produces different, if not completely opposite patterns of abundance. It is understood that fishery independent data are not available in all systems, so landings data are often the only way to obtain some proxy for abundance. However, in systems where effort data are available, using landings concurrently with fishing effort often helps to address some of the potential complications of interpreting fishery dependent data.

Browder's conceptual model suggested that if it is the distance of marsh edge rather than the extent of marsh that drives brown shrimp (*Farfantepenaeus aztecus*) production in LA, then abundance (landings) data should show a strong relationship with the edge (Browder et al. 1989, Zimmerman et al. 2002). This conceptual model was first challenged in this study by pointing out that using landings data to conduct abundance and habitat analyses can be misleading (see above). Chapter 3 found that when brown shrimp survey, landings and landings CPUE data were regressed on the linear distance of marsh edge, differing patterns of abundance occurred. These findings suggest again that using landings data in LA habitat analyses can produce results that may not be reflected in the ecosystem. Moreover, the strong relationship between brown

shrimp and marsh edge interface was not observed when brown shrimp survey data were regressed on the linear distance of marsh edge. Further, other marsh dependent species (white shrimp (*Litopenaeus setiferus*) and blue crab) did not exhibit a significant relationship with marsh edge. These findings suggest that the edge effect, historically thought to drive fisheries production in light of habitat loss, may not be having the significant effect in nature as once hypothesized. While previous studies showed that densities of nekton at the marsh edge are greater when compared with open water habitats, this study contends that studies at smaller spatial scales cannot be extended basin-wide (Minello and Zimmerman 1991, Minello et al. 1994, Zimmerman et al. 2002, Rozas et al. 2007). It is not to say that some members of the population do not find refuge or increased foraging opportunities at the marsh edge, however, it is unlikely that a majority of the population reaches the marsh edge to reap those benefits. Therefore, I contend that the abundance of marsh associated nekton are able to adapt to the loss of habitat, find benefits from other environmental factors not considered in this analysis, or both (Lee 2004, Fry 2008).

5.4. Policy Implications

This research is meant to inform resource managers to better guide the decision making process for restoration and management of both fisheries and marsh habitats. Conservation measures could be better guided if the influence of other factors, such as shallow open bays, contribute more to fisheries production than once thought (Fry 2008). In addition, this study highly encourages the use of the available fisheries independent data collected by the State of LA, as landings data seem to provide a different understanding of how estuarine nekton abundance varies through time. State stock assessments are currently being updated, but much work is still to be done. Until managers have a better understanding of the status of stock for

each important species, analyses in this study could help guide management decisions in the absence of revised stock assessments.

5.5. Limitations and Future Research

Disentangling the influence of environmental drivers from fishing drivers in marine systems has been historically difficult and this study encountered these same challenges. While Ecopath with Ecosim (EwE) has the capability of forcing models with fishing pressure, those data were not included in this study, even though this approach would be a way to account for the confounding factors of environmental and fishing variables. As a matter of coincidence, the edge time series was highly correlated with fishing mortality and fishing effort data. While these correlations might point to the relationship between these factors, the changes in fishing pressure occurred due to outside influences, not related to loss of habitat, such as rises in fuel prices, competition with Asian shrimp markets, and the decrease in fishing infrastructure after Hurricanes Katrina and Rita.

The results of this study were only from one basin within LA. While the analyses from all chapters show similar patterns and the conclusions are therefore considered robust, caution must be taken when expanding these patterns to larger regional scales. It is well known that patterns of land loss and salinity differ widely from basin to basin within LA, so it will be important moving forward to conduct these analyses in other areas to determine if the same relationships hold true.

While Ecosim provides a way to investigate multiple drivers within the ecosystem, it is not capable of showing changes spatially. Addressing habitat loss effects on estuarine nekton using spatially explicit models would be a clear next step. Moving forward, Ecospace, the

spatial portion of the EwE module, will be used to explore the influence of habitat degradation on fish and shellfish in estuaries. Incorporating the latest innovations, historical abundance of estuarine nekton can be investigated in both space and time, while testing differing hypotheses of habitat degradation effects on marine taxa (Steenbeek et al. 2013).

5.6. Closing Remarks

Louisiana's wetland loss is of great concern for more reasons than fisheries production. Wetlands function to improve water quality, protect inland areas from storm surges and flooding, help to prevent shoreline erosion, in addition to providing habitat for numerous other forms of wildlife such as birds, mammals and amphibians. While the influence of marsh loss on fisheries may be less significant than once thought, the importance of protecting coastal wetlands remains vital to the health and prosperity to both the ecosystem and the people that utilize coastal marshes for both recreation and economic benefit.

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